THE HYDROLOGICAL FATE OF NUTRIENTS AND PESTICIDES IN THE URBAN LANDSCAPE IN RESPONSE TO MANAGEMENT AND LAWN SPECIES

by Mark Ryan Slavens

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THE HYDROLOGICAL FATE OF NUTRIENTS AND PESTICIDES IN THE URBAN LANDSCAPE IN REPONSE TO MANAGEMENT AND LAWN SPECIES

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The applications of fertilizers and pesticides to home lawns may contribute to the deterioration of ground and surface water quality. Loss of nutrients and pesticides to ground and surface water can be detrimental for human consumption, reduce recreational usage, and negatively impact aquatic organisms. A field study was initiated to examine the effect fertilizer and pesticide applications can have on the concentrations and mass loss of nutrients and pesticides in leachate and runoff. Studies were arranged on a slope of 13-15% of Arkport Sandy Loam soil (coarse-loamy, mixed, active, mesic Lamellic Hapludalf) through establishing Kentucky bluegrass (*Poa pratensis* L.) sod, or mixtures of Kentucky bluegrass sod and traditional weedy species (*Taraxacum officinale, Trifolium repens, Plantago major, and Digitaria ischaemum*) found in the Northeast of the U.S. Runoff and leachate collected from natural precipitation events were analyzed for nitrate (NO_3^--N), ammonium (NH_4^+-N), total Kjeldahl nitrogen (TKN), phosphate (PO_4^{3-}-P), total Kjeldahl phosphorus (TKP), and applied pesticides. Nitrate contributions in leachate reflected the overall largest potential loss from the landscape. Differences were observed among lawn types for NO_3^--N loss; however significant losses were seen from lawn types regardless of the application of supplemental fertilizer. Ammonium

and PO_4^{3-}-P losses in runoff were significantly higher for the fertilized turfgrass plots compared to non-fertilized lawn types. Although differences in nutrient losses occurred from lawn types, less than 2% of total applied N was lost in leachate and less than 1% and 0.5% of N and P applied were lost in runoff respectively. Pesticide applications could pose a risk to water quality, but the risk to off-site surface water contamination appears to be much more prevalent. Over-time, levels of pesticides in runoff were reduced; however during establishment, concentrations of 2,4-D and Mecoprop were greater than 2800 and 1600 $\mu g\ L^{-1}$, respectively, which could negatively impact aquatic organisms. Overall, differences in nutrient loading existed, but the ability of turfgrass to reduce runoff volumes and utilize applied nutrients appeared significant. The percent of nutrients contributed by fertilizer applications to ground and surface water was minimal and may not pose a significant environmental concern.

BIOGRAPHICAL SKETCH

Born and raised in Wyoming in 1977, Mark R. Slavens attended Jackson Hole High School in Jackson, WY. After graduating, he took a position at Teton Pines Resort and Country Club where he developed a passion for turfgrass science. In 2001 Mark was married to his wife Melanie, and in 2003, Mark graduated from Utah State University with a B.S. degree in Horticulture and Business. Mark received a M.S. degree in Plant Science in 2006 from the same institution under the direction of Dr. Paul Johnson. His continued interests in turfgrass and soil science led him to a research program with Dr. Marty Petrovic at Cornell University where he is completing his Doctoral of Philosophy.

Mark currently is the proud father of two sons, Mason and Porter. With his family, he enjoys all things outdoors including, cycling, running, snowboarding, and golfing.

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LIST OF ABBREVIATIONS

Analysis of Variance (ANOVA)

Days After Seeding (DAS)

Environmental Protection Agency (EPA)

Maximum Contaminant Level (MCL)

Volumetric Water Content (VWC)

Soil Organic Matter (SOM)

Evapotranspiration (ET)

Infiltration Rate (IR)

Honestly Significant Difference (HSD

High Pressure Liquid Chromatography (HPLC)

Gas Chromatography - Electron Capture Detection (GC/ECD)

Cation Exchange Capacity (CEC)

Saturated Hydraulic Conductivity (K_{sat})

LITERATURE REVIEW

Home lawns and turf areas are often maintained in an aesthetic manner that can require fertilizers, pesticides, and supplemental irrigation to maintain healthy and competitive growth dependent on the environmental pressures. The application of nutrients and pesticides beyond what is necessary for optimum turfgrass growth raises environmental concerns that the excessive use of inputs on turf will impact the water quality of rivers, streams, lakes, and other bodies of water.

Non-point pollution has become the most common form of pollution to waterways in the United States making many of them unsuitable for human consumption and recreation (USEPA, 2000). Examples of non-point pollution sources are; construction sites, agricultural fields, residential lawns, and golf courses. Urban non-point runoff is now considered the second most common pollutant to estuaries, third most important cause of lake deterioration and fourth most common pollutant to rivers in the U.S. (USEPA, 2006), and as pollutants continue to contaminate waterways, environmental risks can become problematic and difficult to reverse.

Nitrogen (N) and phosphorus (P) are the two most limiting nutrients to algae in terrestrial systems, and the introduction of non-point pollution can increase the risk of eutrophication. Phosphorus pollution tends to be more of a concern inland when freshwater is present, while N will often be the limiting factor in saline water near coastal environments. As concentrations of limiting nutrients rise, algae can bloom and begin to deplete waterways of oxygen ultimately starving fish and other marine life. The following literature review compiles research results directly related to the role of agriculture and turfgrass pollution to waterways.

Nutrient and pesticide loss from turfgrasses has become a concern in scientific research. The application of nutrients and pesticides can have several fates in turfgrass, but the loss to waterways is a common path. Research in agriculture has shown that nutrients may runoff site through both surface runoff and underground leaching, and dissolved nutrients (<0.45μm) are generally more environmentally damaging than particulate nutrients because they can be transported further in the water system and are more bioavailable (Mathers et al., 2007). There is now evidence to suggest that P from agriculture sources can represent a large input to fresh water, and the increase in P concentrations in agricultural drainage water over time reflects the accumulation of P in soils (Sharpley et al, 2000). Partially due to findings like this and the frequent use of pesticides and fertilizers on golf courses and home lawns, assumptions can be made that turfgrass is a large contributor to non-point pollution of waterways in the United States. It is not appropriate to use results from agricultural studies that monitor leaching and surface runoff and extrapolate them to turfgrass due to differences in management practices, surface mat (thatch), root systems, and plant canopies (Kenna, 1995).

The primary concern associated with turfgrass is the loss of N, P, and pesticides that may move off site and pollute water resources. Nonpoint pollution contributors have become one of the largest inputs of N and P to U.S. waterways (U.S. EPA, 2000) and inputs can be continuous, but are often intermittent and linked to agriculture, or events of heavy precipitation (Carpenter et al., 1998). Eutrophication caused by excessive inputs of P and/or N is one of the most common impairment of surface waters in the U.S. (U.S. EPA, 2000) with impairment measured as the area of surface water not suitable for designated uses such as drinking, irrigation, industry, recreation, or fishing. For nutrient exports to pose water quality risk, there must be a source of the nutrient, mobilization of that nutrient into water, and transportation of that nutrient to a location where its adverse impact is expressed (Mathers et al., 2007).

Nitrogen is an important component in all plant growth, and can be a limiting factor in algae growth in fresh and coastal waters. Atmospheric loss, surface runoff and deep percolation losses of applied N can reduce availability to the growing vegetation, contribute to economic loss to the homeowner, and have potential adverse environmental impacts, therefore, the obvious goal when fertilizing any landscape is to immobilize fertilizer N in the landscape system through plant uptake and soil storage (Erickson et al, 2001). Whenever N is applied beyond plant use and soil storage capacity, it has the potential to be lost to the environment through volatilization, leaching, runoff, or mineralization. With a decrease in the amount of reactive C in the soil, the nitrate absorptive capacity of the soil becomes saturated and concentrations in soil solution and groundwater rise, potentially leading to adverse effects such as eutrophication of terrestrial, freshwater and marine habits, acidification of sensitive freshwaters, and reduced drinking water quality (Rowe et al., 2006).

Phosphorus is considered to be the second most limiting nutrient in waterways for algal growth and is usually a larger concern in freshwater systems. Eutrophication can occur with dissolved reactive P concentrations of 0.01 mg L^{-1} and total phosphorus concentrations of 0.02 mg L^{-1} (Sharpley and Rekolainen, 1997). Soil erosion and nutrient and pesticide loss are important factors when protecting waterways, and erosion and runoff dynamics are complicated. Most erosion in watersheds has been shown to come from a few large storms (Edwards and Owens, 1991). Turfgrass is often used as an erosion control method and in a study of two natural and four manmade erosion control materials, Krenitsky et al. (1998) found that turfgrass sod consistently had the lowest runoff rates and sod reduced runoff by an average of 60% over bare soil, making turfgrass an important tool in erosion control.

Another concern encountered in agriculture and turfgrass management is the potential for pesticides to enter waterways. Pesticides can enter surface water through surface transport and groundwater through leaching (Smith and Bridges, 1996; Ma et al., 1999). The use of pesticides on turfgrass has a tremendous impact on productivity and quality by reducing the impacts weeds, insects, and disease have on turf color, uniformity and density (Smith and Bridges, 1996). Approximately 27 x 10^6 kg of active pesticide ingredients are applied to turf annually in the U.S. to control damage caused by pests (Petrovic and Larsson-Kovach, 1996). Cole et al. (1997) summarized several factors that can affect pesticide and nutrient loss including (i) time interval between chemical application and precipitation event; (ii) amount and duration of precipitation event; (iii) antecedent soil moisture; (iv) slope; (v) amount and method of chemical application; (vi) timing of chemical application in regard to plant uptake; (vii) chemical properties; (viii) rate of field degradation/transformation; (ix) soil properties; (x) and vegetation type or density.

Turfgrass often receives scrutiny for its environmental record, and water quality near golf courses, sports complexes, and large residential sites is questionable. Cohen et al. (1999) conducted a study of water quality impacts of golf courses and found no conclusive evidence that turf management on golf courses was causing significant impacts on ground water quality and that prior and adjacent land use as well as surface and ground water hydrology should be considered for an appropriate analysis. The pathway of nutrient and pesticide loss can be complicated and related to several environmental and physical factors. Predicting water movement in soil and on surfaces involves several variables that make it a multifaceted process.

Nitrogen Fate

Nitrogen is an essential plant nutrient involved in several physiological processes. Most turfgrass managers will base fertility programs on N-rates necessary to provide optimum growth, color, and stress tolerance. Nitrogen applied to a landscape can have several fates including: volatilization, ammonification, nitrification, denitrification, mineralization, and surface and leaching losses. Nitrogen is considered to be highly mobile in the soil and can change forms quickly. All fates are a concern, some effect plant growth, some have environmental implications, and some fates are economically important.

The largest environmental concern of N applications to the landscape is the potential loss of N into either surface or subsurface waterways. Nitrate pollution poses direct health concerns to humans and other mammals (Carpenter et al., 1998) and the EPA has established maximum contaminant levels (MCLs) of NO_3^--N in drinking water of 10 mg L^{-1} to protect babies less than 3-6 months of age (Carpenter et al., 1998). High levels of N in waterways, especially salt water, can produce algal blooms that can deprive the waters of oxygen. Although the primary focus of research in turfgrass relates to waterways, gaseous losses of N_2O (nitrous oxide) through denitrification can be problematic because it is considered a greenhouse gas. Horgan et al. (2002) found labeled fertilizer N (LFN) losses as N_2 and N_2O were 19% for turfgrass compared with 7% for bare soil. Miltner et al. (1996) recovered 64 to 81% of LFN from Kentucky bluegrass with only 0.23% of LFN recovered in leachate, therefore suggesting large volatile losses of nitrogen.

Large quantities of N are necessary for plant health and growth, and reducing losses is environmentally and economically important. One third or more of N applied as a crop can be lost if urea is leached too deeply into the soil by rainfall or irrigation (Mathers et al., 2007). In a study of water quality impacts by golf courses, Cohen et al. (1999) found that out of 906 entries for NO_3^--N in leachate, there were no detections exceeding the 10 mg L-1 MCL and 201 entries in which NO_3^--N was undetectable.

Several factors appear to affect the environmental fate of N in turfgrass including N uptake by the turf (Miltner et al., 1996), rooting characteristics (Bowman et al., 1998; Bowman et al, 2002), irrigation or precipitation rate (Snyder, et al., 1984; Morton et al., 1988; Barton et al, 2006) and application timing (Petrovic, 1990; Easton and Petrovic, 2004; Barton and Colmer, 2006). The establishment period of turfgrass appears to be a critical time for N loss in the turfgrass system. Easton and Petrovic (2004) found NO_3^--N losses were on average two to five times higher during establishment than following establishment of turfgrass.

Phosphorus Fate

Phosphorus is an essential plant nutrient with functions necessary for plant growth. It is one of the primary molecules involved in energy storage and transfer within the plant (ADP and ATP), important in the formation of phospholipids, found in genetic material (DNA) (Corell, 1998), and important for root growth and development (Christians, 2004). In turfgrass management, P is applied through fertilizers or through organic materials such as composts or manures. Phosphorus may be chemically or enzymatically hydrolyzed to orthophosphate, which is the only form of P that can be assimilated by bacteria, algae and plants (Corell, 1998). The plant available form is pH dependent with $H_2PO_4^-$ being dominant at low pH (< 7.2) and HPO_4^{-2} dominates at a higher a pH (> 7.2) (Havlin et al., 2005). Phosphorus is considered to be immobile in soil (Soldat and Petrovic, 2008), and residual P availability can persist for years dependent on soil type, and vegetative removal. In turfgrass, P deficiency symptoms are rarely observed in part due to the fibrous root system of turfgrass that is very effective at mining and taking up available P from the soil. Many turfgrass management programs have begun to promote lower P application rates to reduce the residual amounts of P found in the soil that can potentially have an environmental impact to waterways if lost through surface or subsurface flow.

The loss of P to waterways can have severe environmental impacts. Phosphorus in water is not directly toxic to humans and animals therefore; no EPA drinking water standards have been established. Phosphorus is considered to be a limiting nutrient to algae primarily in freshwater bodies and when P is transported off-site to lakes, rivers, estuaries and wetlands; the potential exists for eutrophication due to algal blooms, starving the water of oxygen necessary to support life. Soluble P is bioavailable for algal uptake and tends to be the focus of the majority of research pertaining to phosphorus. For most lakes, streams, reservoirs, and estuaries, concentrations of 100 μg total P L^{-1} are unacceptably high and concentrations of 20 μg L^{-1} are often a problem (Corell, 1998). Phosphorus can be transported to waterways either through soluble or particulate forms bound to sediment (Carpenter et al., 1998; Sharpley et al., 2000), and the amount of P lost to surface waters tends to increase with the P content of the soil (Carpenter et al., 1998). Dissolved nutrients are generally more damaging to the environment than particulate forms because they can be transported further into waterways and tend to be more bioavailable (Mathers et al., 2007). Sharpley et al. (1992) found the percent bioavailability of particulate P transported in runoff from watersheds decreased with an increase in sediment concentrations of runoff. One should not assume that particulate forms of P or dissolved organic P are inert in waterways because under appropriate conditions, these forms can be converted to dissolved orthophosphate (Corell, 1998). Knowing that fine particles are the major source of waterway pollution, grass is very effective for sediment control (Deletic, 2005). Particulate P associated with sediment and organic matter may comprise a long-term source of bioavailable P in waterways (Sharpley et al., 1992). Phosphatase enzymes catalyze the hydrolysis of labile forms of organic P to release P as inorganic orthophosphate (Mathers et al., 2007).

With few exceptions, surface waters receive most of their P in surface flows rather than in groundwater since phosphates bind to most soils and sediment (Corell, 1998). Barton et al. (2006) found the total P losses from Bermudagrass leachate were low (< 4 kg P ha^{-1} after 22 months) representing less than 1.5% of the applied phosphorus. King et al. (2006) found dissolved reactive P concentrations in the subsurface drainage water under turfgrass were greater than concentrations measured in tile drains from agriculture. Shuman (2004) found the mass of both NO_3^--N and P transported to waterways from turf is directly correlated to the rate of application, suggesting small applications of fertilizers in order to reduce nutrient losses. Sediment-bound or particulate P includes P associated with soil particles and large molecular-weight organic matter eroded during flow events and constitutes the major proportion of P (60-90%) transported form most cultivated lands (Sharpley et al., 2000). Soluble P transport is dependent on desorption-dissolution reactions releasing P from the soil, fertilizer products, and vegetative type while bioavailable particulate P is a function of physical processes controlling soil loss and particle-size enrichment and chemical properties of the eroded soil material (Sharpley et al., 1992).

In an overview of P loss in agriculture, Sharpley et al. (2000) summarizes that surface runoff from grassland, forest land, and or non erosive soils carries little sediment and is therefore generally dominated by dissolved P. In turfgrass and prairie plots soluble P comprises the majority of P in runoff due to relatively dense vegetative cover in buffer strips which inhibits soil erosion and corresponding particulate P (Steinke et al., 2007). In grasslands, Watson et al. (2007) found a highly significant ($p < 0.001$) increase in annual Olsen-P status in plots receiving P fertilizer, which increased with the rate applied. In the plots where no fertilizer P was applied, there was a small, but significant ($p < 0.001$) annual decrease in Olsen-P. Easton and Petrovic (2004) found that regardless of fertilizer source, the highest nutrient concentrations in surface flow from turfgrass occurred in the first runoff event following application, which was generally with 20 days. Shuman (2002) also found a major flush of fertilizer P occurs with initial rainfall events after P applications, and very little fertilizer P is transported in later rainfall events. Although turfgrass may receive P fertilization, it may not pose a greater risk to environmental surface water quality than non-fertilized environments. Ultimately, even low maintenance landscapes can be a source of P in urban environments due to natural and necessary P uptake by leaves and subsequent leaching from vegetation (Steinke et al., 2007).

Pesticide Fate

Pesticides are commonly applied to turfgrass in order to provide dense, healthy foliage and root systems that can provide conditions that appropriate for the user of the site. With the increase use of pesticides due to higher aesthetic requirements and increased pressures placed on turfgrass, there comes the potential risk for pesticides to harm the surrounding environment. Pesticide fate is studied extensively prior to registration, but certain growing conditions such as; turf type, mowing height, irrigation frequency and soil type can affect mobility and transport of pesticides off-site from turfgrass.

Pesticide mobility in surface and subsurface flow depends primarily on two factors: water solubility, and soil sorption coefficient (K_{oc}). These two values are inversely related so when water solubility increases, the K_{oc} decreases and pesticides will desorb from soil particles more easily by the influx of soil moisture. If a pesticide has high water solubility or a low K_{oc} value, there is an increased risk of waterway contamination. A third factor potentially affecting waterway pollution of pesticides is the persistence in the soil. Pesticides that have a half life of less than 30 days are considered to be non-persistent, half lives greater than 30 and less than 100 days are moderately persistent and greater than 100 days is considered persistent. The longer a pesticide persists in the environments, the increased chance that it may move to a waterway or off-site simply because it is available longer to move off-site. A pesticide with high water solubility, low K_{oc} value, and is persistent in the soil would run a higher risk of either leaching or running off site through surface and subsurface flows.

The application of pesticides can affect the environment in other ways besides through contaminating waterways. In a study of lawns, Cheng et al. (2008) found that professional lawn care provided the highest aesthetic lawn quality over do-it-yourself (DIY) and no-input lawn care programs due to better weed control, however, professional programs and DIY programs negatively affected microbial biomass and soil organic matter pools.

Similar to runoff and leaching of N and P, most often the highest concentrations of herbicides in runoff and leachate occur during the first rain event following application. Smith and Bridges (1996) found that concentrations of mecoprop in a greenhouse lysimeters study only occurred in the leachate for 21d after treatments were applied. For all herbicides (2, 4-D; mecoprop, dicamba), the total analyte transported from the greenhouse lysimeters was less than 1.0% of the herbicide applied. The highest concentrations of herbicides found in leachate did not exceed 6 $\mu g\ L^{-1}$ and the concentrations of dicamba in the leachate from bentgrass did not exceed the minimum detectable level. The highest quantities of 2, 4-D transported from lysimeters containing bentgrass ranged from 3.5 and 5.8 $\mu g\ L^{-1}$ dependent on formulation. The MCL of 2, 4-D of 70 $\mu g\ L^{-1}$ was at least 10 times above the highest concentration determined for the herbicides transported from the lysimeters studies. In a simulated runoff study on bermudagrass, over 75% of all the herbicides collected in runoff were transported from the plots during the first rainfall event, and only samples collected over the first 192 hours after treatment contained concentrations above the minimum detectable levels. The concentration of 2, 4-D in the runoff water was a factor above the recommended MCL of 70 $\mu g\ L^{-1}$ which suggest the potential for waterway contamination and environmental implications.

Cisar and Snyder (1996) studied the mobility of pesticides to a USGA green and found that less than 0.1% of organophosphate pesticides applied were recovered in percolate water, regardless of variations in rainfall and total percolation. They concluded that most of the pesticide was retained in the thatch layer and degraded by microbes over time. Chlorpyrifos leaching followed a similar trend when only 0.2% of the total was lost during the first application and only 0.08% was lost during the second application even though the percolation volume increased fivefold. Leaching of isazophos and isofenphos applications resulted in less than 0.1% of total applied for both application dates as well showing the relatively low mobility of some pesticides even in a sand based root-zone where infiltration and percolation rates are high.

Runoff and leaching losses of pesticides in turfgrass appear to be relatively small. Haith and Rossi (2003) found mean annual pesticide runoff loads did not exceed 3% of the annual applications for any pesticide or site, and applications applied to moss were substantially less than 1% of the application. The risk to surface and subsurface waterways appears to be small when pesticides are applied to turfgrass at appropriate rates and large precipitation events do not directly follow the application dates.

Nutrient and Pesticide Pathways

Leaching

The process of leaching water and subsequent nutrients and pesticides from soil is dependent on several factors. Leaching is the process of soil moisture draining out of the root zone with the potential for movement of water and nutrients into aquifers, water tables, and other waterways. The rate of leaching is related to both the infiltration rate and the hydraulic conductivity of the soil. Generally coarser-textured soils containing macro pore space are more prone to leaching, while finer textured soils can hold more water resulting in less leaching losses, but potentially larger runoff losses. A large contributor to leaching losses is preferential flow which is created through cracks in the soil caused by earthworm holes, roots, frost, or other similar processes. Channels are created for water to flow through allowing for faster transport of nutrients and pesticides that may ultimately end up in waterways (Kung et al, 2000).

When looking at home lawns or other turfgrass sites, there are several factors that may affect leaching and ensuing nutrient and pesticide losses. In general, flux rates of nutrients to water from fertilizer or other sources are influenced by the rate of application, season, vegetative cover, chemical form, method of application, soil moisture, irrigation, and the amount and timing of rainfall after application (Petrovic, 1990; Carpenter et al., 1998; Easton and Petrovic, 2004; Barton et al., 2006). Nitrogen is often the most common concern for leaching losses due to its properties and high mobility in soils, but the amount of NO_3^--N leaching can be highly variable and range from 0 to 81% of that applied to turfgrass (Petrovic, 1990). These values are similar to losses of N found in agricultural systems where percentages of fertilizers lost to water range from 10 to 40% for loam and clay soils and 25 to 80% for sandy soils (Howarth et al., 1996). A well maintained turf has been shown to intercept and immobilize ^{15}N-labeled nitrogen quickly making leaching an uncommon avenue of N loss from the turf system and fertilizing Kentucky bluegrass in late fall poses very little potential for significant groundwater contamination where turf density and organic matter are high (Miltner et al, 1996).

In regard to rate and method of application as it pertains to N losses through leaching, several factors are important. The chemical form of N being applied can reduce the amount of N lost through the soil profile (Barton and Colmer, 2006). The use of organic forms, or sulfur coated urea has been shown to reduce NO_3^--N losses in leachate over the use of NH_4NO_3 and urea forms (Snyder et al., 1984; Petrovic, 1990) and the return of clippings can increase NO_3^--N concentration and mass loss in percolating soil water (Kopp and Guillard, 2005). Bowman et al (1998) found leaching of NH_4^+-N was retarded by increased cation exchange capacity and more rapid uptake by the turfgrass plant and that NO_3^--N uptake was inhibited by an average of 70% by the presence of NH_4+ potentially increasing risk of NO_3^--N leaching.

The method of applying nutrients to turfgrass is important and usually done in either a liquid or granular form. Nutrients already in soluble form can potentially be leached if irrigation or large rain events follow application while granular forms would be most likely lost to surface flow if large precipitation events follow application. Homeowners and turfgrass managers apply fertilizers at various rates and at various frequencies. To reduce costs of fuel and labor, fertilizers can be applied at higher rates and applied less frequently. Typically a slow or controlled release fertilizer would be used to elongate the efficacy of the nutrients between applications. Using a smaller application rate more frequently can reduce the risk of both leaching and runoff losses of nutrients (Easton and Petrovic, 2004; Barton and Colmer, 2006; Frank et al., 2006) by applying a frequency that closely matches the plants needs (Barton et al., 2006).

Vegetative cover is important in the landscape and can influence leaching. Increased plant density can create a more convoluted pathway for surface flow, causing it to slow and allowing more time for infiltration into the soil (Linde et al., 1998; Deletic, 2001; Easton and Petrovic, 2004; Pan et al, 2006). The different surface architecture of various turfgrasses compared to other weedy broadleaf species may have an effect on both runoff and leaching of nutrients and pesticides.

Vegetative cover is also critical in determining leaching and runoff dynamics related to plant rooting. The establishment period of turfgrass presents the highest risk to ground and surface water quality and fertilizing at this time poses the greatest risk to water quality due to soil compaction caused during establishment that may reduce infiltration rates and the lack of rooting (Easton and Petrovic, 2004). Bowman et al.

(2002) found that that NO_3^--N and NH_4^+-N readily leached from all species of turfgrass studied, but declined substantially as the root system developed. During establishment, the soil organic matter can be reduced, and turfgrass management practices that decrease soil carbon (C) sequestration may be expected to increase N mineralization and N leaching potential (Barton and Colmer et al., 2006) especially in areas where the soil has been disturbed (Miltner et al., 1996).

Rooting depth and structure is important in affecting both preferential flow and infiltration rates of a soil. Turfgrasses tend to have a fibrous root system that supports soil structure and reduces runoff volumes (Krenitsky et al. 1998). If large portions of broadleaf weedy species are present in a turf stand, there may be the potential for increased nutrient and pesticide losses based of changes in the root architecture of the landscape. If plants have deep taproots that form pathways or create preferential flow for water movement, downward flow may increase and nutrients and pesticides may enter waterways more readily.

In a study of six different warm season turfgrasses, Bowman et al. (2002) found that St. Augustinegrass was the most efficient and zoysiagrass the least efficient species at reducing N leaching. The study also concluded that St. Augustinegrass had nearly twice the root mass of any other species, potentially increasing its ability to absorb NO_3^--N from the soil system. Bowman et al. (1998) found NO_3^--N concentrations in leachate under shallow rooting creeping bentgrass being two times higher (38.2% of total applied N) than those of deep rooting creeping bentgrass plants (18.5% of total applied N). Increased root density may reduce N leaching by providing more root interception of N, or may increase soil microorganism populations that immobilize N (Bowman et al., 2002). Horgan et al. (2002) studied nitrate losses in Kentucky bluegrass turf versus bare soil and found that bare soil plots contained four times as much LFN in leachate than on turfgrass plots when irrigation frequency and amount were the same for both systems.

Growing season is an important factor in nutrient and pesticide losses through leaching. Cool season turfgrass plants follow a seasonal pattern and shoot and root development is the largest in the spring and the fall and shoot and root dieback can occur in the summer season. Cool season turfgrasses will enter dormancy in the winter months and no longer take up nutrients or pesticides that may still be stored in the soil. If applications of nutrients and pesticides are made too late in the year when the plant is entering a dormant state, they may not be taken up and the risk of environmental losses can increase. Petrovic et al. (1986) found that if a soluble form of urea is used to fertilize in the late fall, a significant amount of NO_3^--N leaching can occur in turfgrass. Application of nutrients when the plant is actively growing can be an effective management strategy to reduce nutrient and pesticide loss through plant interception with an increased root architecture, and increased uptake due to plant needs and increased microbial activity. King et al. (2006) detected a strong seasonal pattern in N leaching; NO_3^--N was present in greater concentrations in subsurface drainage water during the winter months than during the spring and summer months when the turfgrass was actively taking up N and the greatest dissolved reactive P concentrations in subsurface drainage water were measured during the fall.

Irrigation and rainfall have a significant role in nutrient and pesticide fate in turfgrass. Increased irrigation rate and increased N rate can enhance the potential for leaching losses from turfgrass (Morton et al., 1984; Snyder et al., 1984; Bowman et al., 1998; Kopp and Guillard, 2005; Barton et al., 2006). In a study conducted by Barton et al. (2006) on sod production, nitrate concentrations in leachate samples ranged from 0 to 69 mg N L^{-1} with the highest concentration reported for the water-soluble fertilizer when applied at the highest rate (300 kg N ha^{-1} per crop) with the highest irrigation treatment. Contradictory to findings for nitrate, the highest ammonium concentrations leached in those plots were from water-soluble fertilizer at the highest application rate at the lowest irrigation treatment. In this study, increasing the irrigation treatment significantly increased the N leached from on average 44 to 132 kg N ha^{-1}. Under the high irrigation treatment, the greatest N losses occurred during the establishment of the turfgrass, and were reduced to a lesser extent at all other times. When irrigation is delayed 3 to 5 days after N applications, both shallow and deep rooted genotypes of creeping bentgrass were able to absorb most of the applied NO_3^--N as evident by the dramatic reduction in N loss (Bowman et al. 1998).

There is large variability in the way leaching will occur on turfgrass sites and N losses are a great concern. Cohen et al. (1999) conducted a study on water quality impacts of thirty six golf courses and found no maximum contaminant level (MCL) (10 mg L^{-1}) exceedances of NO_3^--N in surface water and only 3.6% of ground water samples exceeded the MCL. It was determined that most exceedances were due to prior use of the land before the golf course. After 748 days post-treatment, LFN applied to Kentucky bluegrass turf totaled 0.18% of the amount applied (Miltner et al., 1996). King et al. (2006) found the greatest concentrations of NO_3^--N measured in leachate occurred seven to nine months after application and that the timing of NO_3^--N movement through subsurface drainage from turf appeared to be dependent on climactic factors. The process of subsurface flow and nutrient losses is complex, and difficult to predict based on the many factors that can influence the pathway.

Runoff

Runoff or overland water-flow is a significant source of off-site nutrient loss that poses a threat to waterways. Runoff can be caused by natural precipitation events or through irrigation events to a landscape or turf area. Irrigation and precipitation runoff occurs through two primary processes: (i) through irrigation and precipitation excess runoff when water is applied a surface at a rate higher than that surface's infiltration rate (Cole et al, 1997; Walter et al., 2003; Shuman, 2004); and (ii) soil saturation runoff which occurs when the soils pore space is completely saturated and cannot hold water until some has been displaced (Walter et al., 2003; Shuman, 2004; Srinivasan et al., 2007), either through plant uptake, evaporative losses, or leaching losses.

Several factors affect runoff dynamics and the volume of water, sediment, nutrients, and pesticides that travel off-site from turfgrass. The major components shown to influence runoff volumes and frequency in the landscape include: soil physical properties, site slope, vegetative cover, season, precipitation or irrigation rate, and antecedent soil moisture content. In most cases, surface waters receive most of their P through surface flows rather than through groundwater since phosphorus binds strongly to most soil and sediment (Corell, 1998). Therefore, the focus of nutrient losses in runoff tends to be on P while N losses are more strongly associated with leaching loss. Erickson et al. (2001) found N losses from surface runoff were insignificant from St. Augustinegrass and an alternative residential landscape with 10% slope and frequently intense rainstorms in a sandy soil. Exceptions to this idea would include plants grown on sandy soils, some organic soils, and soils with large percentages of macropores that could allow for losses of P through subsurface flow (Mathers et al., 2007) or when precipitation events follow shortly after fertilization applications and N travels in surface flow.

Soil physical properties are critical in predicting not only runoff volumes, but potential contaminants that may travel in the runoff. In general, the coarser soils containing larger portions of sand and silt are less prone to runoff loss, but may be more prone to leaching loss. This is due to a higher infiltration capacity, reducing the frequency of irrigation or precipitation excess runoff. Finer textured soils have lower infiltration capacities and can experience excess runoff more quickly which can cause golf course fairway runoff to vastly exceed runoff from greens due to the enhanced infiltration on the sand-based greens (Haith and Rossi, 2003). Soil type has also shown relationship with runoff volumes through surface sealing. Whenever soil is disturbed, it is critical to provide surface cover to minimize raindrop impact energy. This force breaks up soil aggregates and detaches soil particles, causing soil particles to become suspended in runoff (Krenitsky et al., 1998). These smaller particles of clay can then clog larger pore spaces on the surface of the soil, reducing infiltration rates, causing runoff to occur more quickly (Gross et al., 1991; Carroll et al., 2000).

The slope of a turfgrass site is going to directly influence whether precipitation or irrigation is lost through overland flow or subsurface flow. As much as 70% of rainfall can occur as runoff water from sloped areas (Smith and Bridges, 1996). Vegetative cover can reduce both erosion and runoff losses when compared to bare soil (Krenitsky et al., 1998; Carroll et al., 2000; Pan and Shangguan, 2006; Pan et al. 2006).

Vegetative cover is important to reduce raindrop impact energy which can dislodge sediment from the surface (Krenitsky et al., 1998; Pan et al., 2006) carrying with it sediment bound nutrients and pesticides. The increase in plant density can reduce the hydrodynamic power of flowing water and allow for water to infiltrate into the ground (Linde et al., 1998; Deletic, 2001; Pan et al., 2006). Pan and Shangguan (2006) found that grass cover reduced runoff rates by about 14-25% compared to bare soil, and reduced sediment yield by 81.2 to 94.3%. They attributed a large portion of this to an approximately 50% reduction in flow velocity in grass plots compared to bare soil plots, although there is good evidence that the reduced raindrop impact energy was the primary difference in erosion (Heilig et al., 2001; Gao et al., 2005). Grass has a very high roughness that slows the approaching overland flow allowing time for water to infiltrate (Deletic, 2001).

Research has been conducted on the density of turfgrass, turfgrass height, turfgrass type, and thatch content to determine their effects on runoff. The dense growth habit of turfgrass and associated organic matter help retain water compared to other urban and agricultural land uses (Haith and Rossi, 2003). Linde et al (1995) studied variation in runoff among creeping bentgrass and perennial ryegrass and found as grasses matured, runoff from perennial ryegrass occurred earlier and at greater volumes than from bentgrass. It was determined that the high-density, thatch forming creeping bentgrass produced a more tortuous pathway that increased hydraulic resistance and in turn increased surface residence time which allowed for greatest total water infiltration on the bentgrass plots. This study did not take into effect evapotranspiration losses and the antecedent moisture content of the soil. A second study by Linde et al. (1998) found that creeping bentgrass reduced runoff volumes over the less dense, lower thatch forming perennial ryegrass when keeping the antecedent moisture content constant among the turfgrasses thus supporting the findings in the first study.

Easton and Petrovic (2004) studied fertilizer source and the effect on runoff in turfgrass. Fertilization of the turfgrass created higher shoot density and higher soil infiltration rates, lower soil moisture levels and subsequently less runoff. The lower runoff volumes achieved through fertilization generally reduced nutrient losses in runoff, and as the shoot density doubled, the infiltration rate increased, reducing runoff by threefold. Cole et al. (1997) studied waterway buffers and their effect on runoff from bermudagrass and found that mowing height reduced runoff losses from buffers mowed at 1.3 cm compared with 3.8 cm and attributed the runoff losses to increased turf density attained by shorter mowing height. Treatments in this study that contained buffers were effective in reducing losses of all pesticides, NH_4^+-N, and PO_4^--P in runoff water compared to plots with no buffers. Steinke et al. (2007) found that turfgrass plots reduced runoff losses by 54% compared to prairie plots in 2004 and that total P losses from prairie were four times greater than turf during that same growing season.

Turfgrass cover has been shown to reduce runoff volume and the transport of NO_3-N and P through a reduction of soil moisture prior to rainfall events and that rainfall-runoff dynamics were closely related to initial moisture conditions (Shuman, 2004). Given the high evapotranspiration (ET) and infiltration rates of most turfgrass systems, runoff volumes may be insufficient to wash significant amounts of chemicals off-site (Haith, 2001). Runoff events typically occur in response to intense rainstorms, and runoff volumes are greater when the antecedent soil moisture content is higher (Linde et al., 1995). Reduced runoff volumes in creeping bentgrass can partially be attributed to higher ET losses and lower antecedent moisture contents over perennial ryegrass (Linde et al., 1995). Runoff losses can be directly related to the initial soil moisture content prior to irrigation or precipitation events and if the soil is allowed to dry, runoff will be delayed and larger precipitation events will be needed to cause runoff loss (Srinivasan et al., 2007). If initial soil moisture content is high, saturation excess runoff may occur from even small rainfall events that typically would not produce runoff.

The season of the year is an important factor when maintaining turfgrass plots. Fertilizers or pesticides are applied during seasons that have been shown to be more prone to runoff and leaching losses. Generally, runoff and leaching losses are lowest when the turfgrass is actively growing and taking up water and nutrients from the soil. As discussed previously, when a turfgrass plant has a deep, well characterized root system, it is able to intercept more water and nutrients potentially reducing leaching losses. This function also helps to reduce the moisture content of the soil and in return reduce runoff losses by reducing the chance for saturation excess runoff. For cool season grasses, optimum conditions for shoot and root growth occur in the spring and fall when soil temperatures are adequate and above ground temperatures are low enough that drought and heat stress is not as prevalent (Christians, 2004). During the peak of the summer, ET losses are typically at their highest, and dependent on weather conditions can range close to 6 mm d^{-1}. In theory, these high ET losses will reduce soil moisture content and provide more pore space in the soil for infiltration of precipitation when events occur, reducing runoff and leaching losses. On the other hand, in the summer months, root die back and turf thinning is a common response to heat and drought stress which can provide less root interception of water and nutrients and a less dense turf canopy to reduce runoff and sediment volumes.

Winter can be an important time for runoff losses, especially if the soil freezes and forms an impermeable surface for precipitation and snowmelt. Easton and Petrovic (2004) found in the second year of a two year study that snowmelt or runoff on frozen soils accounted for 62% of total runoff collected over the entire year for Kentucky bluegrass and perennial ryegrass plots. Steinke et al. (2007) found a direct relationship between runoff volume and season and on an annual basis, the effect of vegetation type was negligible as it had no effect on reducing runoff during the winter. Over 80% of the annual runoff occurred during the winter because the frozen ground conditions caused vegetated surfaces to act as an impervious surface. Sediment loss from the prairie plots tended to be the greatest when the foliage was dead or dormant resulting in reduced aboveground biomass. Watson et al. (2007) found there is a significant ($p<0.001$) seasonal variation in dissolved reactive P concentrations found in runoff for grassland plots, with higher concentrations occurring between May and September compared with the rest of the year. Seasonal variation may not only affect runoff volumes, but may affect nutrient and chemical loads in runoff, therefore, it is important to have a full understanding of potential risks associated with runoff losses for all times of the year and not just when the plants are actively growing.

Sediment loss in runoff is a major concern due to the possibility of nutrients and chemicals bound to the sediment that may enter waterways. Without vegetative cover, the risk of runoff and sediment loss can greatly increase due to rainfall impact energy, lower infiltration rates, higher antecedent moisture contents, and increased water velocity. Sediment transport mainly depends on runoff flow rate (velocity and depth), grass length, slope, density of plants, and sediment size and density (Deletic, 2005). In a 28 year study involving nine watersheds, Edwards and Owens (1991) found that the three largest erosion-producing storms on each watershed accounted for more than 50% of the soil loss measured in a period. Gross et al. (1991) found different seeding rates of tall fescue had a statistically significant impact on soil loss and that for three rainfall intensities, sediment loss rates were highest at the 0 kg ha^{-1} seeding rate compared to 98, 244, 390, and 488 kg ha^{-1} seeding rates. Deletic (2005) found that concentrations of sediment in runoff decreases exponentially with the distance down the plot, where smaller particles travel much further than coarse ones, and the first 30 cm of downhill turf trapped the largest amount of sediment. Pan et al. (2006) found that plots containing a combination of turfgrass and moss reduced sediment loss by 45.2% to 84.5% over no grass and moss plots.

The application of nutrients and pesticides to turfgrass for aesthetic purposes can be defined in this study as “cosmetic applications”, in which no benefit to plant growth and development is received from these applications beyond what is aesthetically pleasing to the user. The cosmetic application of fertilizers and pesticides is constantly under scrutiny and several counties and states in the U.S. are considering banning the cosmetic applications of such products. This study plans to address the potential relationship of cosmetic applications of fertilizers and pesticides to turfgrass to nutrient and pesticide losses in runoff and leachate.

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CHAPTER 1

THE FATE OF NITROGEN IN THE URBAN LANDSCAPE IN RESPONSE TO LAWN SPECIES COMPOSITION AND MANAGEMENT

Abstract

The application of fertilizers and pesticides to home lawns offers the potential to contaminate ground water, causing it to be unfit for human consumption. However, the application of fertilizers and pesticides for cosmetic purposes not only can improve the aesthetics of a home lawn, but can improve turfgrass shoot and root density, water use, organic matter accumulation, and other factors known to be influential in urban hydrology. Therefore, a field study was set up to determine the impact fertilizer and pesticide applications to home lawns contribute to the concentration and mass loss of three forms of nitrogen in leachate and runoff compared to various types of non-fertilized lawns. Free draining lysimeters were place at the top of a slope in a sandy loam soil (coarse-loamy, mixed, active, mesic Lamellic Hapludalf) to monitor the total volume and nutrients and pesticides lost in free draining leachate. Leachate and runoff collected from natural precipitation events were analyzed for nitrate (NO_3^--N), ammonium (NH_4^+-N), total Kjeldahl nitrogen (TKN). Nutrient concentrations fluctuated over-time from all lawn types and regardless of fertilizer inputs, NO_3^--N concentrations from all lawn types were above the EPA drinking water standard of 10 mg L^{-1} at some point in time during the 27 month study. However, when looking at the cumulative mass of nutrients in leachate, NO_3^--N loads were significantly impacted by the first 6 sampling periods, especially the winter precipitation in the first year of the study in which the majority of the contributions were seen. There were significant differences observed among treatments; the most drastic of which was the reduced level of loading from lawn predominately populated with broadleaf weeds. After accounting for controls, total N loss from the plots in leachate was <2% of N

applied. Ammonium was the largest portion of N lost in runoff, and concentrations measured were directly related to fertilization timing. However, similar NH_4^+ loads were seen from all plots, regardless of fertilization applications.

Home lawns and turf areas are often maintained with applications of fertilizers, pesticides, and supplemental irrigation to uphold healthy and competitive growth and high visual appeal. The application of nutrients and pesticides to lawns and turfgrass areas for cosmetic purposes raises environmental concerns, and the potential of contaminating ground and surface waters with nutrients and chemicals does exist. Off-site nutrient losses can negatively impact the water quality of rivers, streams, lakes, estuaries, and groundwater for both human consumption and recreation, and for aquatic organisms. Non-point pollution has become the most common form of pollution to waterways in the United States making many of them unsuitable for human consumption and recreation (USEPA, 2000). Urban non-point runoff is now considered the second most common pollutant to estuaries, third most important cause of lake deterioration and fourth most common pollutant to rivers in the U.S. (USEPA, 2006). As pollutants continue to contaminate waterways, environmental risks can become problematic and difficult to reverse.

Most turfgrass managers or do it yourself practitioners will base fertility programs on the nitrogen (N) rate necessary to provide optimum growth, color, and stress tolerance. The N applied to a landscape can have several fates including: volatilization, ammonification, nitrification, denitrification, mineralization, and surface runoff and leaching losses (Petrovic, 1990; Miltner et al, 1996; Horgan et al, 2002). The largest known environmental concern of N applications to the landscape is the potential loss of N into either surface or ground water. Nitrate pollution poses direct health concerns to humans and other mammals (Carpenter et al., 1998) and the EPA has established maximum contaminant levels (MCLs) of NO_3^--N in drinking water of 10 mg L^{-1} to protect babies less than 3-6 months of age (Carpenter et al., 1998). High levels of N in waterways, especially salt water estuaries, can produce algal blooms that can deprive the waters of oxygen. Although the primary focus of research in turfgrass relates to surface and ground water, gaseous losses of N_2O (nitrous oxide) through denitrification can be problematic because it is considered a greenhouse gas. Horgan et al. (2002) found labeled fertilizer N (LFN) losses as N_2 and N_2O were 19% for turfgrass compared with 7% for bare soil. Miltner et al. (1996) recovered 64 to 81% of LFN from Kentucky bluegrass with only 0.23% of LFN recovered in leachate, therefore suggesting large gaseous losses of nitrogen.

When looking at residential lawns or other turfgrass sites, there are several factors that may affect leaching and the ensuing nutrient losses from the site. In general, flux rates of nutrients to groundwater from fertilizer or other sources are influenced by the rate of application, season, vegetative cover, chemical form, method of application, soil moisture, irrigation, and the amount and timing of rainfall or irrigation after application (Petrovic, 1990; Carpenter et al., 1998; Easton and Petrovic, 2004; Barton et al., 2006). Nitrogen is often the most common concern for leaching losses due to its properties and high mobility in soils, but the amount of NO_3^--N leaching can be highly variable and range from 0 to 81% of that applied to turfgrass (Petrovic, 1990). In a 20 year comprehensive study of eighty golf courses, NO_3^--N was detected in 82% of groundwater samples; however, only 1.2% of those samples exceeded the EPA drinking water standard of 10 mg L^{-1} (Baris et al., 2010). A well maintained turf has been shown to intercept and immobilize ^{15}N-labeled nitrogen quickly making leaching an uncommon avenue of N loss from the turf system (Horgan et al., 2002). While some have found that fertilizing Kentucky bluegrass in late fall has been shown to pose little potential for significant groundwater contamination where turf density and organic matter are high (Miltner et al, 1996;

Geron et al., 1993); however, the potential for N to leach during the late fall has been well documented (Magiafico and Guilliard, 2006; Petrovic, 2004). Petrovic (2004) found that if a soluble form of urea is used to fertilize in the late fall, a significant amount (29-47% of the amount applied) of NO_3^--N leaching can occur in turfgrass. Application of nutrients when the plant is actively growing can be an effective management strategy to reduce nutrient and pesticide loss through plant interception with an increased root architecture, and increased uptake due to plant needs and increased microbial activity. King et al. (2006) detected a strong seasonal pattern in N leaching; NO_3^--N was present in greater concentrations in subsurface drainage water during the winter months than during the spring and summer months when the turfgrass was actively taking up N and the greatest dissolved reactive P concentrations in subsurface drainage water were measured during the fall.

Rate and method of application affects N leaching losses. The chemical form of N being applied can reduce the amount of N lost through the soil profile (Barton and Colmer, 2006). The use of organic forms, or sulfur coated urea has been shown to reduce NO_3^--N losses in leachate over the use of NH_4NO_3 and urea forms (Snyder et al., 1984; Petrovic, 2004). Using a smaller application rate more frequently can reduce the risk of both leaching and runoff losses of nutrients (Easton and Petrovic, 2004; Barton and Colmer, 2006; Frank et al., 2006) by applying a frequency that closely matches the plants needs (Barton et al., 2006).

Differences in N leaching exist among species (Lui et al., 1997; Erickson et al., 2010) and can also be dependent on root architecture and development. Reductions in NO_3^--N and NH_4^+-N leaching have been attributed to more extensive root systems in warm season turfgrasses (Bowman et al., 2002) and NO_3^--N concentrations in leachate under shallow rooting creeping bentgrass has been documented as two times higher (38.2% of total applied N) than those of deep rooting creeping bentgrass plants (18.5%

of total applied N)(Bowman et al., 1998). Increased root density may reduce N leaching by providing more root interception of N, or may increase soil microorganism populations which may immobilize N (Bowman et al., 2002). Furthermore, simply having turfgrass cover over bare soil has been shown to reduce LFN in leachate (Horgan et al. 2002).

Age of site affects the extent of nitrogen leaching. During establishment, the soil organic matter can be reduced, and practices that decrease soil carbon (C) sequestration may be expected to increase N mineralization and N leaching potential (Barton and Colmer et al., 2006) especially in areas where the soil has been disturbed (Miltner et al., 1996). King et al. (2006) found the greatest concentrations of NO_3^--N measured in leachate occurred seven to nine months after application and that the timing of NO_3^--N movement through subsurface drainage from turf appeared to be dependent on climactic factors.

Irrigation and rainfall have a significant role in nutrient fate in turfgrass. Applying an increased irrigation rate along with an increased N rate can increase the potential for leaching losses from turfgrass (Morton et al., 1984; Snyder et al., 1984; Bowman et al., 1998; Kopp and Guillard, 2005; Barton et al., 2006). In warm season grass sod production, NO_3^--N concentrations in leachate samples range from 0 to 69 mg N L^{-1} with the highest concentration reported for water-soluble fertilizer applied at the highest rate with the highest irrigation treatment (Barton et al., 2006). The highest NH4+-N concentrations in leachate in those plots were from water-soluble fertilizer at the highest application rate at the lowest irrigation treatment (Barton et al., 2006). The process of subsurface flow and nutrient losses is complex, and difficult to predict based on the many factors that can influence the pathway.

Runoff of nitrogen for turfgrass sites is affected by several factors, and practices that influence density and infiltration rate such as fertilizing (Easton and

Petrovic, 2004; Bierman et al, 2010), reducing mowing height (Cole et al., 1997 that reduce runoff volume and nutrient loss. Nitrogen losses in runoff from turfgrass are often seen as insignificant (Erickson et al. 2001). In 20 years of comprehensive research monitoring surface runoff from golf courses, 1809 samples were collected throughout the U.S. and only 20 samples had NO_3^--N levels above the EPA MCL standard (Baris et al., 2010). Runoff losses can be directly related to the initial soil moisture content prior to irrigation or precipitation events and if the soil is allowed to dry, runoff will be delayed and larger precipitation events will be needed to cause runoff loss (Linde et al., 1995; Srinivasan et al., 2007). Nitrogen found in runoff is lower when the soil moisture content is low prior to runoff (Shuman, 2004). Reduced runoff volumes in creeping bentgrass was partially attributed to higher ET losses and lower antecedent moisture contents compared to perennial ryegrass (Linde et al., 1998). Winter can be an important time for runoff losses, especially if the soil freezes and forms an impermeable surface to precipitation and snowmelt. Direct relationships between ground freezing and runoff volume have been established and the majority of runoff from northern climates can be attributed to frozen conditions (Easton and Petrovic, 2004; Steinke et al., 2007; Bierman et al., 2010). Seasonal variation may not only affect runoff volumes, but may affect nutrient loads in runoff, therefore, it is important to have a full understanding of potential risks associated with runoff losses for all times of the year and not just when the plants are actively growing.

The objective of this study was to examine the effect of perceived cosmetic applications of nutrients and irrigation to newly established turfgrass lawn has on N mobility in surface and ground water, and to monitor the effects these applications have on the density and composition of the species within the plots. These lawns were compared to lawns composed of turfgrass, annual and perennial weedy species that receive no maintenance beyond mowing for loading of nitrogen species to surface and groundwater.

Materials and Methods

The following study was conducted at the Turfgrass and Landscape Research Center at Cornell University in Ithaca, NY. A southwest facing hill-slope having a slope range of 13-15% and measuring 39 m by 6.1 m was stripped of all sod to a depth of 3.8 cm. Surface oil was removed with a bulldozer and stockpiled for later use. The subsoil was removed to a depth of 45 cm and the underground subsoil was thoroughly compacted with the bulldozer by repeatedly driving over the site. The remaining subsoil and surface soil was replaced and compacted in 15 cm lifts with the bulldozer to ensure consistent compaction levels and to provide conditions similar to high soil disturbance of a new home construction site.

The hill slope was subdivided into thirty two 1.2 m wide by x 6.1 m long plots. Lysimeters were constructed similar to Miltner et al. (1996) made of polyvinyl chloride (PVC) pipe with a 61.0 cm diameter and 91.4 cm length. Each length of PVC was driven into the ground vertically in the top 1.2 m section of the hill slope, using a loader until the top of the lysimeters was flush with the surface of the soil. All attempts were made to remove the lysimeters with the soil core intact, and once removed, a 10 cm gravel layer was placed at the bottom, and a flat sheet of 12.7 mm thick PVC was attached to the base with PVC cement to ensure a tight seal. If a soil core was not removed intact, the soil was removed in 15 cm lifts and placed in the lysimeters in the same sequence it was removed, and then tamped to ensure the same quantity of soil fit into the lysimeters to maintain the bulk density of the soil profile. Approximately one half or two replicates of the lysimeters were removed with the core intact and the other half were removed by hand. Once all bases were affixed to the lysimeters, they were lowered back into the excavated site they were removed from.

A trench 1.0 m deep was placed on the downhill slope of the lysimeters to give access to the base and a 12.7 mm diameter exit drain was placed at the bottom of each lysimeters. Polyethylene pipe with a diameter of 12.7 mm was connected to each exit drain with a barbed fitting and tubing from eight lysimeters was run downhill to a common collection area for leachate sampling in 4000 mL plastic containers. After the installation of the lysimeters, trenches were backfilled and compacted to prevent settling. The entire area was covered in approximately 7-8 cm of soil and hand raked in preparation for establishments.

The study consisted of a randomized complete block design with the variation in the slope across the plots used as the blocking factor to provide four blocks with eight treatments per block. Plastic landscape edging was placed between each plot to a depth of 50 mm to confine flow of surface water within plots. Treatments consisted of Kentucky bluegrass (*Poa pratensis* L.) (KB) lawn that received applications of fertilizers (F) and pesticides (P) using consumer lawn products (The Scotts-Miracle Gro Comp., Marysville, OH) that provide 4 fertilizer applications and 4 pesticide applications; pendimethalin [N-(1-ethylpropyl)-3,4-dimethyl-2, 6-dinitrobenzenamine] in spring, mecoprop [2-(4-Chloro-2-methylphenoxy)propionic acid] + 2, 4-D (2,4-dichlorophenoxyacetic acid) in late spring and early fall, and bifenthrin [(2-methyl-1,1-biphenyl-3-y1)-methyl-3-(2-chloro-3,3,3-trifluoro-1-propenyl)-2,2-dimethyl cyclopropanecarboxylate] in summer, and supplemental irrigation (I). Irrigation was applied when soil moisture content decreased below 20% relative water content which was determined to be the level of 50% depletion of plant available water (Brady and Weil, 1999). Plots were irrigated with 1.25 cm of water at each irrigation event with a below ground irrigation system; irrigating at a rate of 75 mm hr^{-1}. The volume of irrigation water applied did not induce runoff losses. The second treatment was the same as the first described above, but without supplemental irrigation. The third treatment consists of KB receiving all four fertilizer applications, but no pesticide and no supplemental irrigation applications. The fourth treatment consists of KB with no applications of pesticides, fertilizers, or supplemental irrigation and was treated as the untreated control plot. The remaining four treatments represented more common weedy lawns found in humid temperate landscapes that receive no fertilizers, pesticides, or irrigation. The fifth treatment was a lawn consisting primarily of typical cool season broadleaf weeds (*Plantago major*, *Taraxacum officinale, and Trifolium repens)* that was only mowed and will be referred to a 100% broadleaf weed lawn. The sixth treatment was a lawn with 50% KB lawn plugged with 50% of the above described broadleaf weeds that was only mowed. The seventh treatment was established entirely as annual grassy weed, and was seeded with smooth crabgrass (*Digitaria ischaemum*.) and will be referred to as 100% crabgrass. The eighth plot consists of mixture of 50% KB and 50% smooth crabgrass that was only mowed. All plots were mowed weekly at a height of 5 cm with a rotary mower similar to those used in a home lawn and all clippings were removed. Plots that received supplemental fertilization received total amount of N equivalent to 360.3 kg ha^{-1} for the entire 27 month study period (Table 1-1).

Plots that contain KB were sodded on 3 July 2007 with sod containing four different cultivars (Odyssey, Liberator, Rugby II, and Total Eclipse). Plots that consist of both 50% broadleaf weeds and 50% crabgrass weeds, had approximately 400 -108 mm diameter plugs of KB removed and plugs of broadleaf plantain (*Plantago major*), white clover (*Trifolium repens*), and common dandelion (*Taraxacum officinale*) were plugged into place while plugs of smooth crabgrass (*Digitaria ischaemum*) were used on the 50% crabgrass plots. All plots were plugged until a 1:1 ratio of weeds to KB was obtained. For plots consisting of primarily broadleaf weeds, sod with approximately 90-100% of the described broadleaf weeds was stripped from a site at the research center and placed on the plots. The 100%

annual grassy weed plots were seeded with smooth crabgrass at a rate of 614 kg ha^{-1} on 3 July 2007 and over seeded annually at a rate of 200 kg ha^{-1} on 15 May 2008 and 12 May 2009. All plots were irrigated to facilitate root establishment of sodded plots and seedling establishment of crabgrass plots for a four week period by keeping the plots well watered.

During establishment, a 155 mm wide by 200 mm deep trench was placed at the bottom of all plots running perpendicular to the hill slope. A 127 mm wide, 1.20 m long channel drain gutter was cut to the length of each plot and placed with a 0.5% slope at the base of each plot with a 12.7 mm diameter drain placed at one end connected to polyethylene pipe that led the water to a collection site for the surface runoff water. All containment gutters were covered with 3.2 mm thick metal sheeting to stop rain and outside contaminants from entering the collection site. The drains for the channel gutters ran downhill where piping was grouped for every two plots collection area where soil was excavated and a 95 L plastic storage container was placed in the soil. The polyethylene pipe entered through the uphill side of the container and was extended over 100 mL tipping buckets. Tipping buckets were leveled and placed on 12.7 mm thick plywood with holes drilled large enough for 1000 mL sampling bottles to be placed below the buckets. Tipping buckets were connected to event data loggers (Onset Computer Corp., Pocasset, MA, model HOBO® pendent event logger) and every tip was calibrated prior to use by pouring 1000 mL of water into each collection gutter periodically and measuring the number of tips for the given volume. The number of tips and frequency of tips were measured for each storm event and logged to equate total runoff losses from each plot. Runoff storm events were monitored and classified as a precipitation event when runoff occurred to any plot within the study.

Precipitation that produced leachate and runoff were separated by precipitation event; however, often multiple storms occurred within several hours to days of each other and leachate volumes could not be separated from events. In such a case, total leachate volumes were recorded for the entire precipitation event and the event was labeled as a single precipitation event with the precipitation amounts recorded for the entire period. Subsamples of leachate and runoff were taken within 6 h of storm events, and frozen immediately until further analysis. Subsamples were compiled into sample periods based on the frequency of storms in a time period to obtain 3-6 storms per period and to reduce the number of samples to analyze. Subsamples were bulked proportionally to the volume of leachate for the given storm event to create bulked subsamples with nutrient concentrations proportionate to the actual storm events. Once bulked, subsamples were filtered through a 45 μm filter to remove sediment and to analyze for nutrients.

Table 1-1. Application dates and product descriptions of the fertilizer and pesticide combinations applied to the Kentucky bluegrass turfgrass plots.

Application date	**Product N-P-K ratio**	**Nitrogen applied**	**Phosphorus applied**	**Potassium applied**	**Active ingredient†**
		kg ha^{-1}			
3 September 2007	28-3-8	36.5	1.7	8.7	Bifenthrin
24 September 2007	26-3-12	30.1	1.5	11.5	2,4-D; Mecoprop
23 April 2008	30-3-4	41.2	1.8	4.6	Pendimethalin
26 May 2008	28-3-3	39.2	1.8	3.5	2,4-D; Mecoprop
28 July 2008	28-3-8	36.5	1.7	8.7	Bifenthrin
3 October 2008	26-3-12	30.1	1.5	11.5	2,4-D; Mecoprop
29 April 2009	30-3-4	41.2	1.8	4.7	Pendimethalin
26 May 2009	28-3-3	39.2	1.8	3.5	2,4-D; Mecoprop
22 July 2009	28-3-8	36.5	1.7	8.7	Bifenthrin
16 September 2009	26-3-12	30.1	1.5	11.5	2,4-D; Mecoprop
Total applied		**360.6**	**16.8**	**76.9**	

†Active ingredients only applied to lawn treatments receiving pesticides

Samples were analyzed for NO_3^--N (EPA method 114-A) by reducing nitrate through a copperized cadmium coil to nitrite and measured colormetrically at 520 nm. Samples were analyzed for NH_4^+-N using (EPA method 351.2) by reacting ammonia with alkaline salicylate and hypochlorite to form an analog of indophenol blue that was measured colorimetrically at 650-660 nm. Total filtered Kjeldahl nitrogen (TKN) was digested according to (EPA method 136-A) under the presence of sulfuric acid, potassium sulfate and copper (II) sulfate to convert organic compounds to either ammonia or orthophosphate. Samples were then analyzed according to the protocols for the given compounds described above in the Turfgrass Research Laboratory at Cornell University Ithaca, NY 14853 using a auto discrete analyzer (Seal-Analytical, Mequon, WI, Model: AQ2).

Plant density was measured monthly through digital photo analysis to determine potential relationships between canopy density and nutrient and stormwater/ leachate losses. A digital camera (Olympus Inc., Model: 5 Megapixel Stylus) was mounted on a monopod that stands 100 cm above the plot and extended out 60 cm horizontally away from the vertical axis to allow the camera to be positioned directly above the plots without obstruction (Richardson et al., 2001). Photos were taken of the top, middle, and bottom portions of the plots. All photos are taken between the 1100 and 1300 h. to optimize light and provide consistent conditions. Photos were analyzed using digital imaging software (SPSS, Inc., Chicago, IL, Sigmascan Pro v. 5.0) by separating hues related to plant color of 45 to 107 and a saturation range from 0 to 100. Hue ranges between 57 and 107 have been shown to selectively identify green leaves in an image (Richardson et al., 2001), but through our analysis, a wider range was found to be more optimum to isolate plant color for this study. Total pixels related to green leaves were then divided by the total pixels in the image to give a percent coverage of the plots.

Volumetric moisture content of plots were taken periodically using time domain reflectometry (TDR) through a water content probe (Spectrum Technologies Inc., Plainfield, IL, model: FieldScout TDR 300) with 12 cm rods to monitor irrigation frequency and antecedent conditions. Measurements were taken at the top, middle, and bottom of each plot to quantify dynamics that may affect water movement above or below the soil surface.

The study was initiated on 10 August, 2007 when the first supplemental irrigation event was applied to KB plots receiving irrigation plus pesticides and fertilizer applications and storm events were monitored for 27 months through 1 November, 2009. During the second winter of the study, frost tubes (McCool and Molnau, 1984) were place in the center of each plot to determine the relationships between cover type and soil freezing, and also the relationship between soil freezing depth and runoff and leaching frequencies. Data was monitored primarily during the initial soil freezing period or when climatic conditions were favorable for snowmelt or rainfall that may induce leaching.

On 2 September 2008, 7.3 cm diameter undisturbed soil cores were taken 6 cm deep from each plot. A soil core was taken from the top, middle, and bottom portion of each plot and stored in a cooler at 4°C. Samples were analyzed for distribution of pore space, penetrometer resistance, and bulk density as described in (Moebius et al, 2007).

Soil cores measuring 2.2 cm diameter x 8 cm deep were extracted on 26 September 2007, 21 October 2008, 26 June 2009 and 12 August 2009 from several random locations within each plot and homogenized to create a soil sample representative of each plot. Samples were used to determine soil particle distribution through the hydrometer method (Sheldrick and Wang, 1993), Morgan extractable phosphorus (Morgan, 1941), soil pH in 0.01 M $CaCl_2$ (Hendershot et al, 1993):

organic matter from loss on ignition (Karam, 1993).

Pre-established soil infiltration potentials were measured within each block on 23 May 2007 using a miniature rain infiltrometer (Ogden et al. 1997) and secondary infiltrations measurements were taken on 5 June 2008 and 23 September 2009 within each plot to monitor changes in infiltration through the establishment period.

Statistical analysis

Several factors were measured that are known to have the potential to influence the water balance of an urban landscape. Plant density, species composition and VWC were measured over-time, which had a significant effect on their values and were subjected to a repeated measures analysis.

All leaching and runoff nutrient data was subjected to analysis of variance using a mixed model in JMP 8.0 (SAS institute, 2009) with repeated measures. Treatment and sample period were treated as fixed effects, while the blocking factor was treated as a random variable in the model. When significant differences were observed, means were separated using Tukey's HSD with $\alpha = 0.05$ probability level.

Soil sample and plot characteristics were subjected to ANOVA with repeated measures with the exception of undisturbed soil core data that was taken only once. If following ANOVA, there was no significant effect of time on a measurement, data for the plot was averaged over time and means were separated using Tukey's HSD test with an $\alpha = 0.05$ probability level when significant effects were observed through ANOVA.

Results and Discussion

Soil properties and lawn type characteristics

Several measurements were taken over the course of the study to develop relationships between lawn types and the effect on hydrologic properties. Table 1-2 summarizes significant different measurements of soil and plot characteristics related to lawn type. Of the measurement summarized, the most distinguishing factors that would affect leaching and runoff dynamics are infiltration capacities (IC, Table 1-2), and volumetric water content (VWC, Fig 1-1) of the lawn types .There was significantly lower infiltration rates associated with the KB that received pesticide applications and the 100% crabgrass plots compared to the two lawn types that contained broadleaf weedy mixtures (Table 1-2). Although differences were observed, IC for all plots were dramatically improved over the pre-establishment infiltration capacities of <1 cm hr^{-1} following construction and are considered common by ASAE standards (Partsch et al., 1993)

Volumetric water content on average was higher for the irrigated KB lawn type than all other lawn types. It is not surprising that by irrigating to natural precipitation that the VWC was higher that non-irrigated lawns. The fertilized non-irrigated lawn types and the 100% crabgrass plots had significantly lower VWC than the remaining lawn types (Fig 1-1). There were differences in VWC of plots dependent on the location of measurement on the hillslope. We observed as others have found (Easton and Petrovic, 2004) a moisture gradient in all turfgrass plots in which the top of the plot was drier than the bottom of the hillslope. This same gradient did not exist for the 100% broadleaf and the crabgrass+turfgrass plots which had no significant differences in antecedent moisture conditions dependent on hillslope location (Fig 1-1).

In the 27 month period of our study, there were 68 leachate producing precipitation events observed in which subsamples were taken for analysis (Table 1-3), and 74 runoff producing precipitation events observed in which subsamples were taken from for analysis (Table 1-4). One of the most important factors in determining the runoff volume from a site is the storm or precipitation volume and intensity compared to the soils natural infiltration and available storage capabilities, respectively. Based on the 27 month duration of the study, there were only five storm events that produced more than 25 mm or rainfall in a 24 hour period. This factor becomes important, because based on two separate measurement dates, the soil infiltration capacities of the plots ranged from a low of 100 mm hr^{-1} to 210 cm hr^{-1} for the highest (Table 1-2). In fact, during the course of the study, there was never a storm event that produced a precipitation rate large enough to force infiltration excess flow or Hortonian flow (Horton 1933; Horton, 1940) from the plots. Therefore, it can be ascertained that the runoff sampled and measured in this study was derived primarily from saturation excess flow conditions (Hewlett and Hibbert, 1963).

When analyzing the cumulative leachate losses from the lawn types, there were significantly different slopes observed from all eight lawn types, where the majority of losses occurred within the first six sample periods of the study (Fig. 1-2 A and B), especially periods 5 and 6 (12 January to 4 April 2008). We observed several freeze thaw cycles during that period that led to large amounts of snowmelt and with rainfall there were over saturated soil conditions. After the establishment period, the remaining leachate inducing precipitation events contributed less to the total leachate losses from the plots (Fig 1-2 A and B). The turfgrass lawns that received fertilizer applications, but no irrigation had lower leachate volumes than the irrigated turfgrass and the non-fertilized turfgrass. These plots also had lower average soil VWC which can be attributed to lack of irrigation and higher percent density than the non fertilized turfgrass (Table 1-2). The only other lawn type that had the same reduction in leachate volume was the 100% broadleaf lawn (Fig 1-2 B) which also maintained a high vegetative density similar to the fertilized turfgrass plots (Table 1-2).

Through multiple regression analysis, there were significant effects of VWC ($p=0.019$) of the plots as well as plant density ($p=0.018$) on the ratio of precipitation observed as leachate from the plots. The overall differences observed among leaching volumes within treatments are representative of overall differences observed in the moisture contents of the plots, as higher antecedent conditions or lower plant densities provided significantly higher leachate ratios to be observed from the plots. However, these differences were most apparent in the first 6 months of the study, and overtime does not appear to be as important.

The amount of soil organic matter was not was not independent of treatment. The plots that had the lowest SOM content (Table 1-2) were also the plots that had the lowest values for vegetative density (Table 1-2), suggesting that less plant material was available for natural recycling and incorporation into the soil surface as an organic matter component. And although not as significant, there was a relationship between OM content and the bulk density measured in which bulk density decreased as OM increased (Table 1-2).

Table 1-2. Soil and plot characteristics observed in eight different lawn types. Bulk density and soil organic matter were averaged from four different sampling dates. Infiltration rates were averaged over two different time points in two separate growing seasons using a sprinkler infiltrometer. Frost depth is the average observed frost depth over the 2008-2009 winter season.

	Bulk density	**Average Seasonal Frost depth**	**Infiltration capacity**	**Average Vegetative Density**	**Soil organic matter**
Treatment	**g cm^{-3}**	**mm**	**mm hr^{-1}**	**m^{3}m^{-3}**	**g kg^{-1}**
Broad + Turfgrass	1.42 ab	70 a	213.2 a	0.764 ab	53.3 abc
Broadleaf	1.40 b	68 ab	219.7 a	0 .783 a	52.6 bc
Crabgrass + Turfgrass	1.35 b	68 ab	174.3 abc	0.747 c	53.9 ab
Crabgrass	1.52 a	69 a	112.5 bc	0.622 e	45.6 d
Turfgrass	1.42 ab	67 ab	158.1 abc	0.691 d	49.2 cd
Turfgrass + F	1.38 b	50 bc	186.6 ab	0.749 bc	53.2 abc
Turfgrass + F + P	1.41 ab	47 c	101.2 c	0.747 c	57.3 a
Turfgrass + F + P + I	1.45 ab	43 bc	116.4 bc	0.759 bc	52.5 bc

†= values followed by the same letter within columns are not significantly different according to Tukey's HSD test ($p < 0.05$), NS = not significant

‡ Sampling dates for Morgan extractable P, SOM, and pH (26 September 2007, 21 October 2008, 26 June 2009, and 12 August 2009)

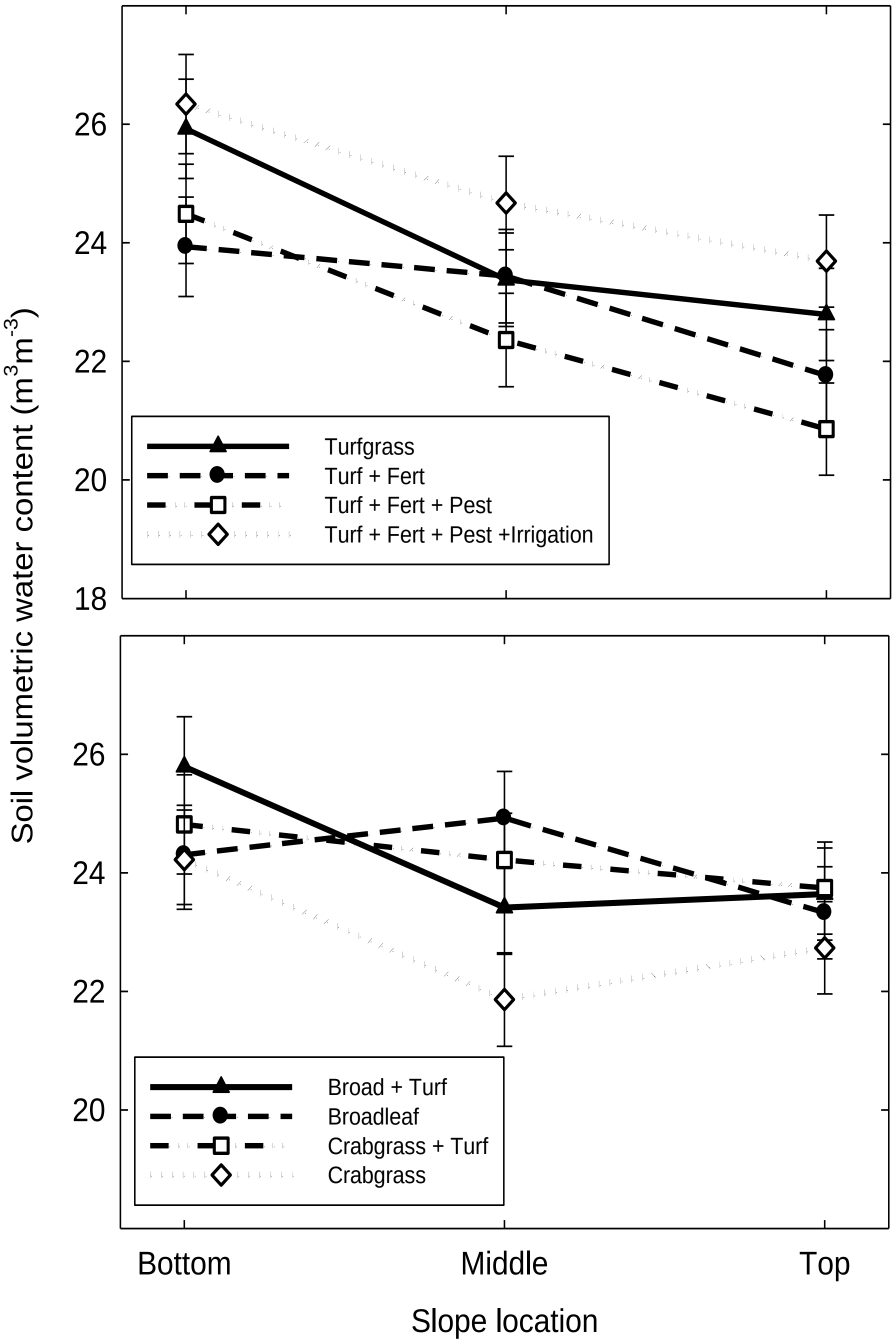

Figure 1-1. Relationships between the soil VWC and location of a hill slope under eight different lawn treatments.

Nitrogen in leachate

Nitrogen species concentrations varied among lawn types but the most striking differences were observed over-time, as noted when differences are observed between the sample period, season, or year (Table 1-5). The treatment consisting of broadleaf and turfgrass planted together had an average NH_4^+-N concentration of 1.24 mg L^{-1} for the 27 months of the study, which was generally 2.5 times higher than any other treatment (Fig 1-2 E and F). Most of this difference can be attributed to sample period 8 (18 May 2008- 15 July 2008), where concentration levels reached 9.2 mg L^{-1} (Figure 1-2 E and F). Most spikes associated with NH_4^+-N in leachate were observed in the winter and spring period of the 27 month study representing the significant interaction among season and year (Table 1-5).

Ammonium-N loading or mass loss in leachate was negligible and, due to low concentrations of NH_4^+-N observed in the samples, the majority of loading was dependent on leachate volume (Fig 1-2 G and H). Similarly to trends in leachate loss over time, nutrient loading was most drastically impacted by sample periods 4, 5 and 6 (especially 12 January to 4 April 2008) that occurred in the late fall through early spring of 2007-2008 (Fig. 1-2 G and H).

Nitrate-N was the form of N that was found in the highest concentration in leachate, regardless of treatment. Similarly to NH_4^+, NO_3^--N loss was significantly affected by the season and year of the study, but there was also a significant lawn type treatment effect observed (Table 1-5). The largest concentrations were observed in the leachate sampling periods 1- 2 (8 Sept- 27- Oct 2007), 9-10 (16 July 2008- 15 November 2008), and 14-15 (7 May 2009- 29 August 2009) (Fig 1-3 E and F).

Regardless of treatment, the leachate samples from all plots had NO_3^--N concentrations that exceeded the EPA drinking water standard of 10 mg L^{-1} (Fig 1-3 E and F). The lowest overall levels of NO_3^--N were seen in the broadleaf, crabgrass, and crabgrass+turfgrass plots, while those consisting of turfgrass, turfgrass+f, turfgrass+f+p+i, and the broadleaf+turfgrass plots had the highest. Overall, the lowest concentrations were seen in the early summer of 2008 (18 May 2008- 15 July 2008) when the plants were actively growing and most likely utilizing soil moisture and available nutrients (Fig 1-3 E and F).

Table 1-3. Leaching events and corresponding date ranges assigned to sample periods

Sample period	**Date range**	**Leachate events**
1	8 September 2007-15 September 2007	1-3
2	16 September 2007-27 October 2007	4-9
3	28 October 2007- 3 December 2007	10-16
4	4 December 2007-11 January 2008	17-23
5	12 January 2008- 29 February 2008	24-30
6	1 March 2008- 4 April 2008	31-37
7	5 April 2008- 17 May 2008	38-39
8	18 May 2008- 15 July 2008	40-41
9	16 July 2008- 10 August 2008	42-44
10	11 August 2008- 15 November 2008	45-47
11	16 November 2008- 5 January 2009	48-51
12	6 January 2009- 30 March 2009	52-55
13	31 March 2009- 6 May 2009	56-57
14	7 May 2009- 30 June 2009	58-61
15	1 July 2009- 29 August 2009	62-64
16	30 August 2009- 15 November 2009	65-68

Table 1-4. Runoff events and corresponding date ranges assigned to sample periods

Sample period	Date range	Runoff events
1	8 September 2007-15 September 2007	1-4
2	16 September 2007-27 October 2007	5-13
3	28 October 2007- 3 December 2007	14-18
4	4 December 2007-11 January 2008	19-21
5	12 January 2008- 29 February 2008	22-28
6	1 March 2008- 4 April 2008	29-33
7	5 April 2008- 17 May 2008	34-35
8	18 May 2008- 15 July 2008	36-40
9	16 July 2008- 10 August 2008	41-45
10	11 August 2008- 15 November 2008	46-50
11	16 November 2008- 5 January 2009	51-53
12	6 January 2009- 30 March 2009	54-57
13	31 March 2009- 18 June 2009	58-62
14	19 June 2009-26 July 2009	63-66
15	27 July 2009- 29 August 2009	67-72
16	30 August 2009- 15 November 2009	73-75

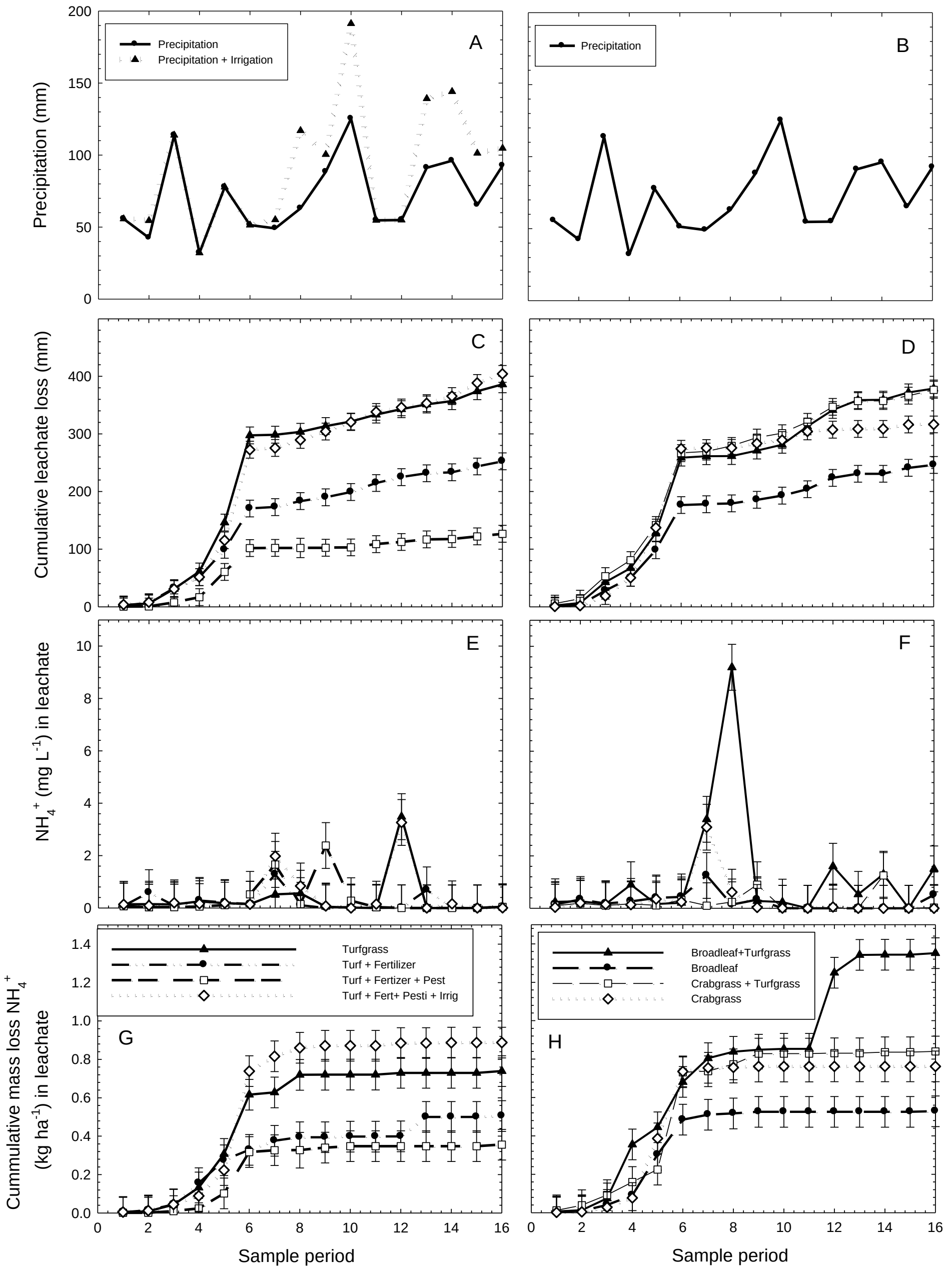

Figure 1-2. Precipitation and irrigation inputs (A)(B), cumulative leachate loss (C)(D), NH_4^+ concentrations (E)(F), and cumulative leaching of NH_4^+ (G)(H) measured from eight different lawn types through natural precipitation events over 16 sample periods over 27 months.

Table 1-5. ANOVA table for nitrogen concentrations (mg L^{-1}) and nitrogen loading (kg ha^{-1}) measured in lawn leachate. ANOVA was performed by multiple regression analysis using a generalized linear model.

		Leachate volume	**NH_4^+**	**NO_3^-**	**Organic N**	**NH_4^+**	**NO_3^-**	**Organic N**
			concentration			**load**		
Source of variation	***df***	***mm***	***mg L^{-1}***			***kg ha^{-1}***		
		p-value						
Lawn type	7	<0.001	ns†	<0.001	ns	0.004	<0.001	0.002
Sample period	1	ns	ns	ns	ns	ns	ns	ns
Season	3	ns	ns	ns	0.005	ns	ns	ns
Year	1	ns	ns	ns	ns	ns	ns	ns
Lawn type*Period	7	ns	ns	ns	ns	ns	ns	ns
Lawn type*Season	21	ns	ns	ns	ns	ns	ns	ns
Lawn type*Year	7	0.003	ns	ns	ns	0.012	0.042	ns
Sample period*Year	1	<0.001	ns	ns	ns	0.004	0.009	ns
Season*Year	3	ns	0.05	<0.001	ns	ns	ns	ns
Lawn type*Period*Year	7	<0.001	ns	ns	ns	0.003	<0.001	0.002

† Indicates non-significant differences at α=0.05 level.

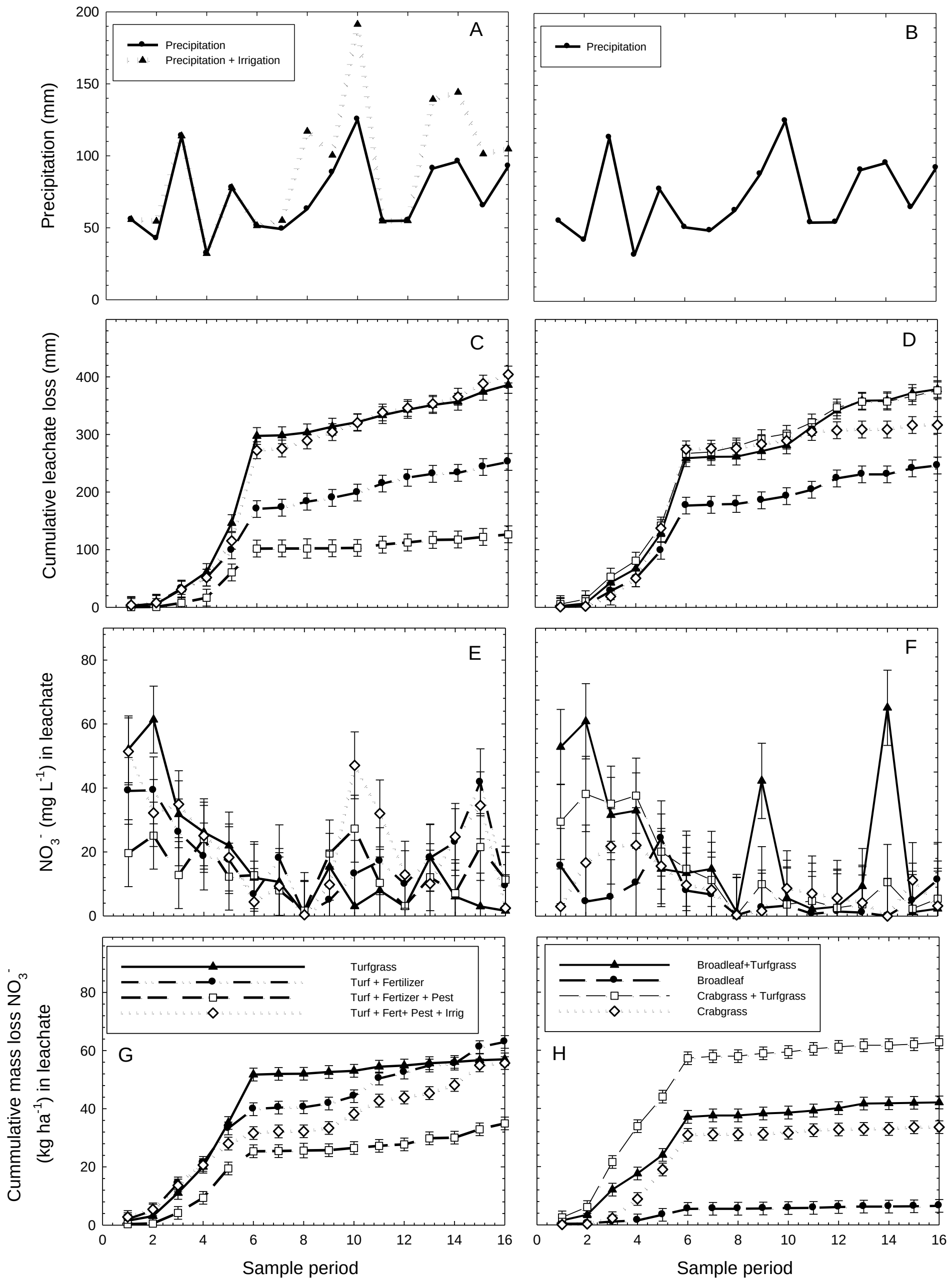

Figure 1-3. Precipitation and irrigation inputs (A)(B), cumulative leachate loss (C)(D), NO_3^- concentrations (E)(F), and cumulative leaching of NO_3^- (G)(H) measured from eight different lawn types through natural precipitation events over 16 sample periods over 27 months.

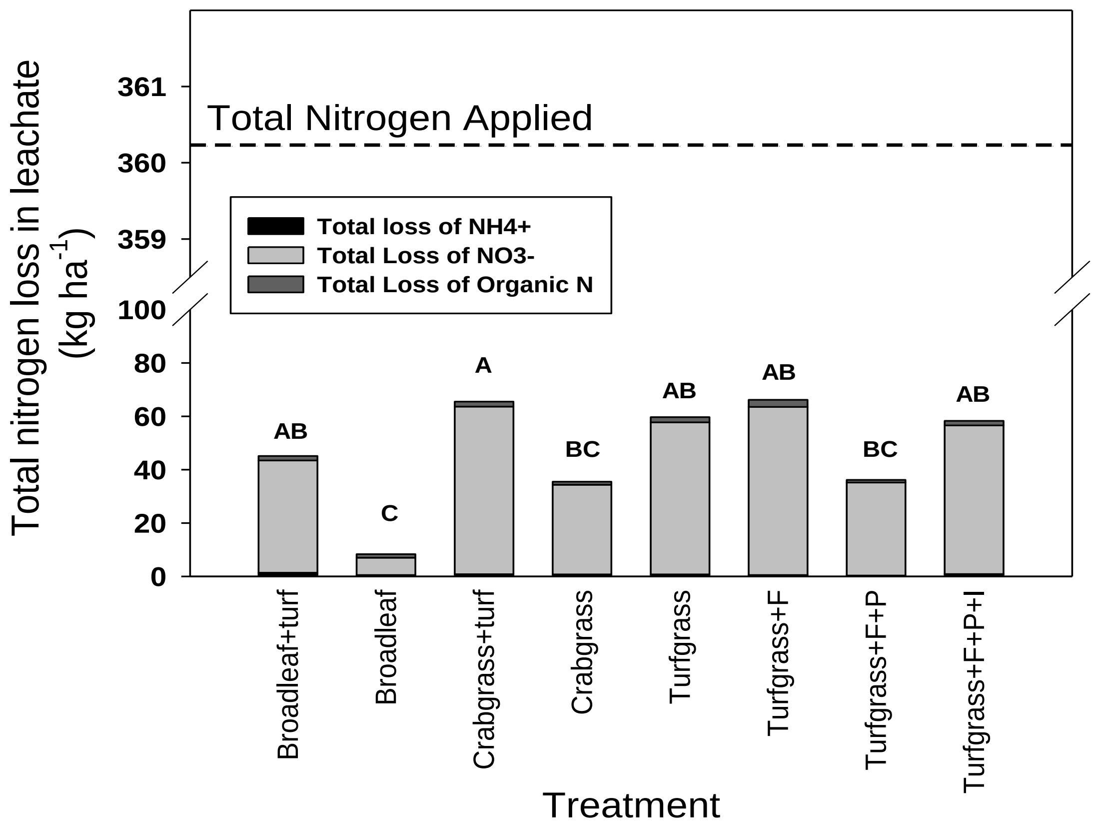

Figure 1-4. Total N leaching loss for the 27 month period for eight different lawn treatments. Treatment bars with different letters represent significant treatment differences ($p < 0.05$)

Nitrate-N loading in leachate was the largest source of nitrogen loss from the plots (Fig 1-4). As seen with NH_4^+-N, the majority of loading occurred in the first 6 months of the study (Fig 1-3 G and H) creating a significant interaction among season and year for NO_3^- loading in leachate. The crabgrass + turfgrass and the unfertilized turfgrass lost 92 and 89% of total loading respectively during the first 6 sample periods while the turfgrass+fertilizer and the irrigated turfgrass loaded 60 and 56% respectively during the same periods. The 100% broadleaf lawn had the lowest total NO_3^--N load, which was significantly lower than several of the lawn types (Fig. 1-3 G and H). The difference in load was related to both low NO_3^--N concentrations in leachate and low leachate loss from the broadleaf plot. Total N loading in leachate from lawn types ranged from 8.3 kg ha^{-1} to 66.15 kg ha^{-1} (Fig 1-4).

Ammonium and organic N contributed minimally to the total N load in leachate during the study (Fig. 1-4). The highest value of total N loss accounted for approximately 1.8% of the total N applied as fertilizer after adjusting for losses from the unfertilized control turfgrass plot, however, differences were not significant among the fertilized and unfertilized turfgrass plots in total N loading in leachate (Fig 1-4). There was however a significant reduction in total N loading in leachate through the use of a predominantly broadleaf weedy plot and values observed under this treatment would most likely be those expected under natural conditions in the N cycle of the labile N pool.

Contributions of organic N to total N concentrations found in leachate were less than 2 mg L^{-1} except for sample period 12 in which there were higher concentrations (Fig 1-5 E and F). Overall, concentrations of organic N were minimal, and the contribution of the organic fraction to total nitrogen loading was a very small component, not more than 2.5 kg ha^{-1} over 27 months (Fig 1-4; 1-5 G and H). In fact, this finding is important because some weedy plots maintained white clover populations of 25-50% over time, and no significant increase in N leaching was observed with the increased input from natural N fixation.

Runoff

The fertilized turfgrass plots that did not receive supplemental irrigation had significantly lower portions of precipitation runoff compared to the 100% crabgrass and the turfgrass plots that received supplemental irrigation (Fig. 1-6 C and D). The only measured difference between the turfgrass lawn types that receive supplemental irrigation and those that do not was the soil VWC. (Fig. 1-1) This factor, along with the difference in soil freezing depth (Table 1-2) which varied among lawn types, were the most important differences observed between the treatments and can help explain the differences seen in the volume of runoff lost from the plots. Freezing depths under fertilized conditions were significantly lower for the winter season, and led to differences in length of the season that the plots were frozen. A shorter winter season with more permeable ground cover observed under fertilized turfgrass reduced runoff volumes over the period.

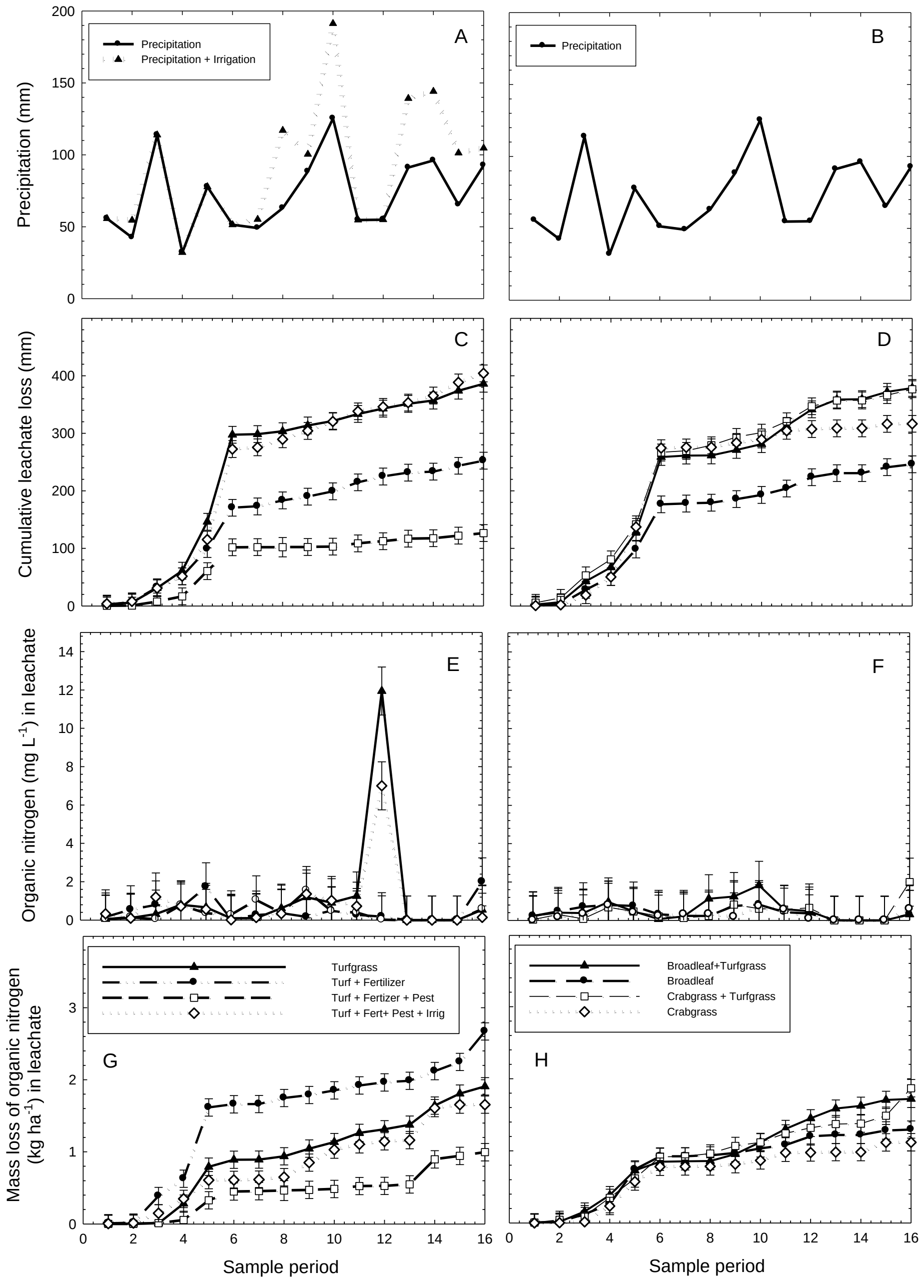

Figure 1-5. Precipitation and irrigation inputs (A)(B), cumulative leachate loss (C)(D), Organic N concentrations (E)(F), and cumulative leaching of Organic N (G)(H) measured from eight different lawn types through natural precipitation events over 16 sample periods over 27 months.

The concentration of in runoff is an important component to overall site loading and the environmental impact. In our study, we observed that high concentrations did not always translate in to greater nutrient loading since the volume of runoff often has a greater effect on nutrient loading (Steinke et al., 2007; Bierman et al., 2010).

Nitrate in runoff

The effect of lawn cover and treatments had on runoff concentration varied dependent on the form of N. Similarly to leachate, seasonality, sample period, and lawn type played important roles in the loss of N in runoff (Table 1-4). The season and year of sampling was the most important factor in determining nutrient concentrations for the three N forms in runoff. There were differences in NH_4^+ concentrations among treatments, and the majority of the spikes in NH_4^+ concentrations seen in runoff for the turfgrass plots were directly related to fertilization timing as observed through differences in fertilized and unfertilized lawn types (Fig 1-6 E and F). The turfgrass + fertilizer plots had significantly higher overall concentrations of NH_4^+ than all other plots except for the turfgrass+fertilizer+pesticide treated plots. This observation strongly supports the potential risk of higher NH_4^+ concentrations in runoff with the applications of fertilizers. However, the use of supplemental irrigation appears to be a mechanism to potentially reduce the higher runoff concentrations of NH_4^+-N as concentrations were reduced under fertilized treatments (Fig 1-6 E and F). Ammonium losses were not solely dependent on treatment, but were also significantly different among sample periods for the lawn types. There was an important trend observed in the weedy mixture plots, in which during the first year of the study, concentrations were consistently low; however, as the plots developed and became more established, we began to observe peaks of NH_4^+ concentrations similar to the fertilized turfgrass plots occurring (Fig 1-6 E and F). This finding is important to the overall N dynamics of the lawn ecosystem considering these plots did not receive any supplemental fertilization.

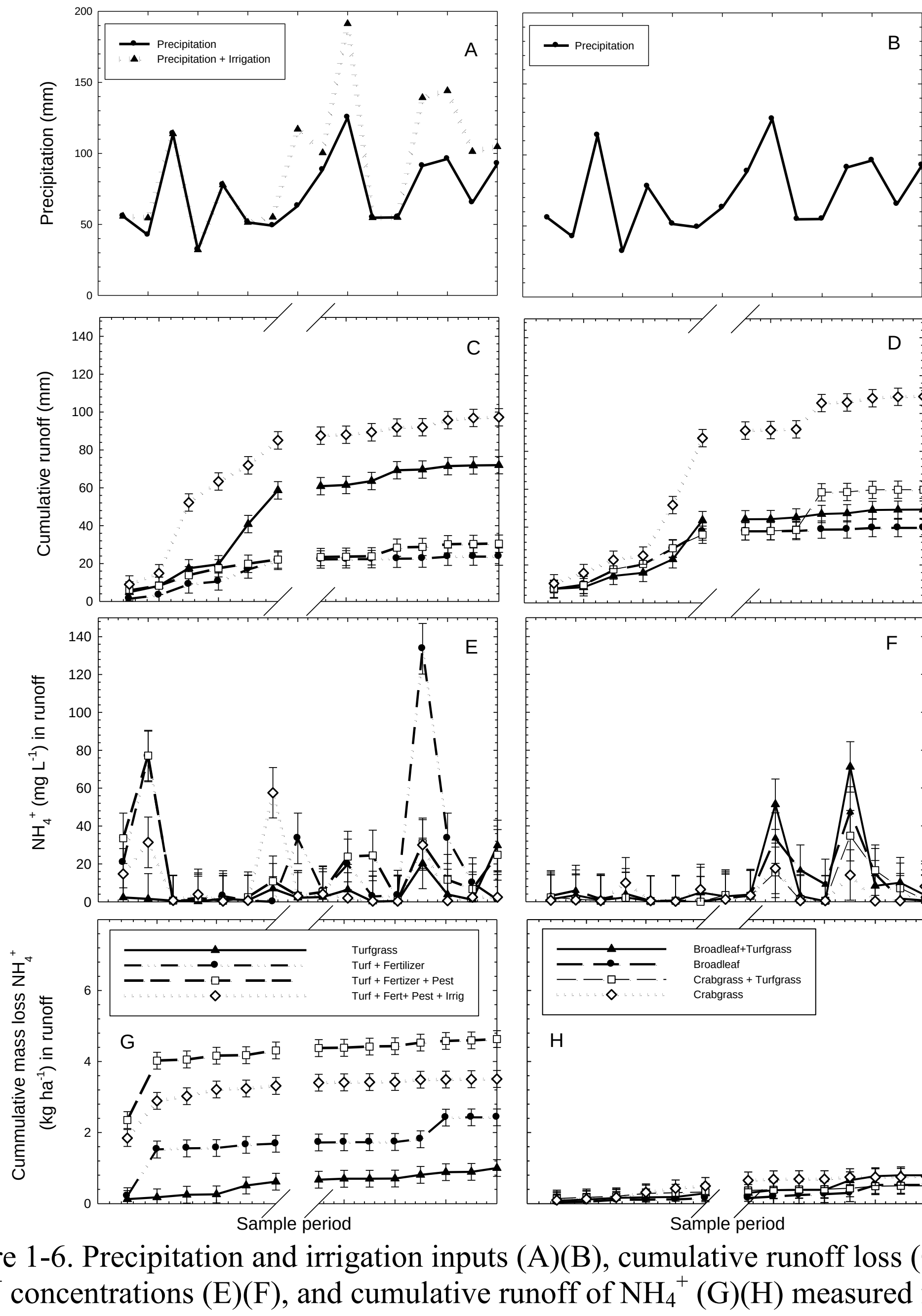

Figure 1-6. Precipitation and irrigation inputs (A)(B), cumulative runoff loss (C)(D), NH_4^+ concentrations (E)(F), and cumulative runoff of NH_4^+ (G)(H) measured from eight different lawn types through natural precipitation events over 16 sample periods over 27 months.

The total loss of NH_4^+ was diminished when converted to from concentrations to total nutrient loading; however differences in loading were apparent among lawn types. The major source of loading of NH_4^+ came from the establishment period of the plots, and obvious effects of fertilization losses were observed as losses occurred for sample periods 1 and 2 from the fertilized turfgrass plots that were not seen under for the unfertilized turfgrass or the weedy plots (Fig 1-6 G and H). All three turfgrass lawn types that received supplemental fertilization had significantly higher NH_4^+ losses compared to the weedy lawn types. The two turfgrass lawn types that received pesticides applications had significantly higher loads than the control turfgrass lawn plots. The application of fertilization was seen to significantly contribute to the total loading of NH_4^+ to runoff compared to the non-fertilized lawn types.

Nitrate concentrations found in runoff were low for all plots however; there were concerning peaks in NO_3^--N concentrations observed. The period when concentrations were the highest was sample period 10, when concentrations in the two broadleaf treatments, the crabgrass+turfgrass treatment, and the turfgrass+fertilizer+pesticide treatments were significantly higher than the other lawn types (Fig 1-7 E and F). Both sample period 10 and 16 led to the highest values of NO_3^- in runoff, and both of these sample periods occurred in the late summer to early fall periods of their respective years (Table 1-4) which appears to be a more susceptible period to observe higher concentrations of NO_3^- in runoff regardless of supplemental fertilizer applications (Fig 1-7 E and F).

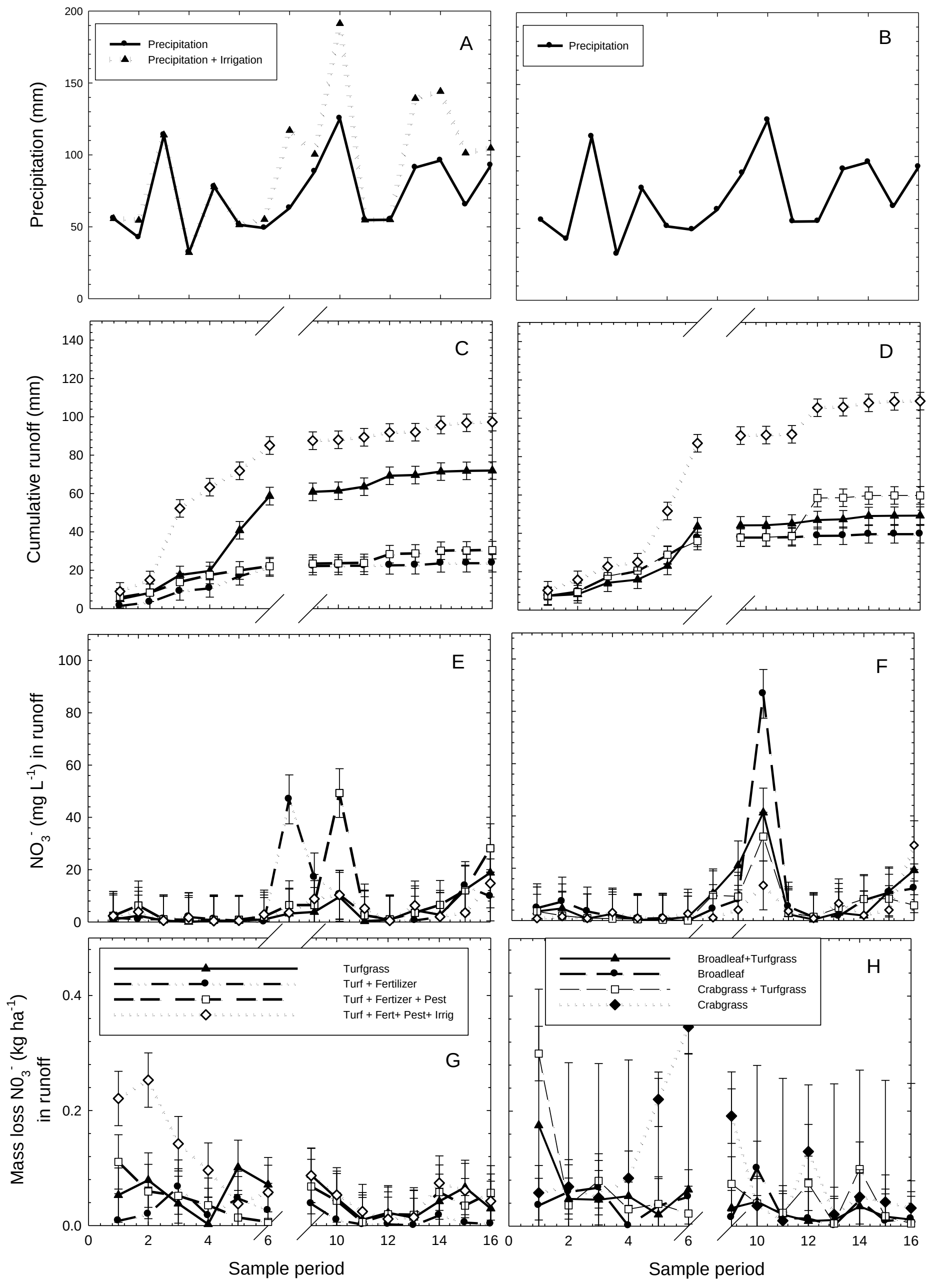

Figure 1-7. Precipitation and irrigation inputs (A)(B), cumulative runoff loss (C)(D), NO_3^- concentrations (E)(F), and cumulative runoff of NO_3^- (G)(H) measured from eight different lawn types through natural precipitation events over 16 sample periods over 27 months.

Mass loss of NO_3^- in runoff was also different among lawn types (Table 1-6). The 100% crabgrass and the irrigated turfgrass lawn types had significantly higher overall loading (Fig 1-7 G and H), and because NO_3^- concentrations in runoff were generally low (Fig. 1-7 E and F), the majority of the loading was caused by the runoff volume differences among the plots. The overall contributions of NO_3^- to the total N load in runoff was a small fraction of the total N load in runoff and does not appear to be a serious pathway for NO_3^--N loss. Similar findings have been seen in a comprehensive review of 20 years of surface water monitoring from golf courses throughout the U.S. (Baris et al., 2010).

Concentrations of TKN found in runoff varied throughout the study, but values were very similar to NH_4^+ concentrations (Fig 1-6 E and F; 1-8 E and F). After analyzing the concentrations of TKN in runoff, ammonium was the largest source of N, and thus contributed the most to the total contribution of total N to the runoff pool. Similarly to NH_4^+, concentrations of total N seen in the unfertilized plots were low for the first year of the study, however, as the plots matured, high levels were observed due to contributions of all N sources to the runoff pool (Fig. 1-8 E and F).

Table 1-6 ANOVA table for nitrogen concentrations (mg L^{-1}) and mass loss (kg ha^{-1}) of nitrogen measured in lawn runoff. ANOVA was performed by multiple regression analysis using a generalized linear model.

		Runoff volume	**NH_4^+**	**NO_3^-**	**Total N**	**NH_4^+**	**NO_3^-**	**Total N**
Source of variation	***df***	***mm***	***Concentration***			***Mass loss***		
			p-value					
Treatment	7	<0.001	<0.001	0.003	<0.001	<0.001	<0.001	<0.001
Sample period	1	ns	ns	ns	ns	ns	0.050	ns
Season	3	ns	<0.001	<0.001	<0.001	ns	ns	ns
Year	1	ns	0.027	ns	0.043	ns	ns	ns
Treatment*Sample period	7	ns	ns	ns	ns	ns	ns	ns
Treatment*Season	21	ns	ns	ns	ns	ns	ns	ns
Treatment*Year	7	ns	ns	ns	ns	ns	ns	ns
Sample period*Year	1	ns	0.021	<0.001	0.002	ns	ns	ns
Season*Year	3	ns	<0.001	<0.001	<0.001	ns	ns	ns
Treatment*Period*Year	7	ns	ns	<0.001	0.036	ns	ns	ns

† Indicates non-significant differences at the α=0.05 level

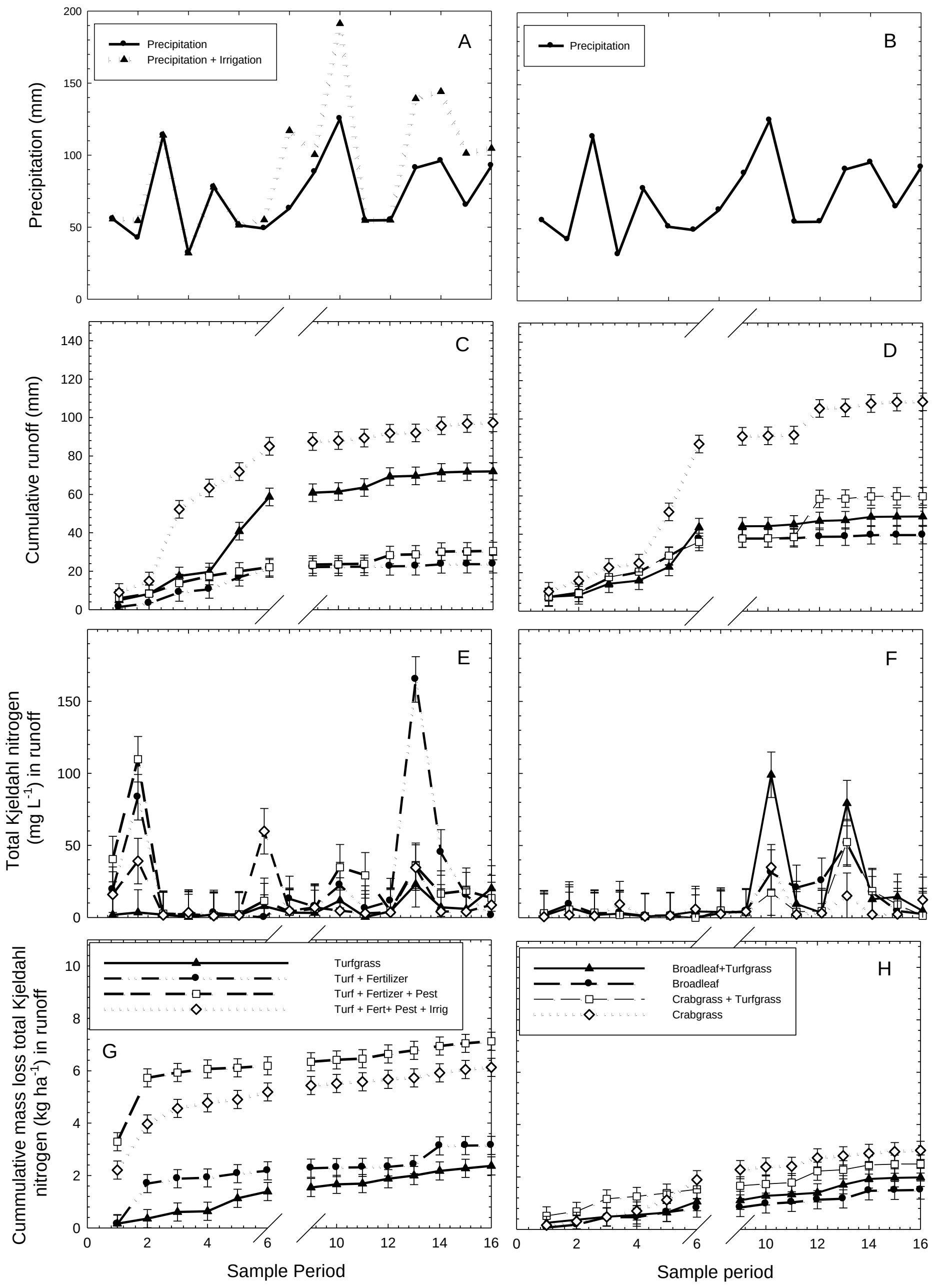

Figure 1-8. Precipitation and irrigation inputs (A)(B), cumulative runoff loss (C)(D), total Kjeldahl N concentrations (E)(F), and cumulative runoff of total Kjeldahl N (G)(H) measured from eight different lawn types through natural precipitation events over 16 sample periods over 27 months.

Conclusion

Minimizing nutrient loss into waterways is an important way to preserve water quality; not only for human consumption and recreation, but for aquatic and terrestrial species that depend on it. Nitrogen losses through leachate and runoff from turfgrass systems can be significant (Petrovic, 1990) and vary widely dependent on species composition and soil and site characteristics. For this study, it became apparent that there were climactic and site conditions that led to the overall loss of leachate and runoff that were the driving factors for N loss. Various differences occurred in conditions that affect the hydrology of a site based on the treatments imposed on the site. The most critical factors that influenced nutrient loss were lawn type differences in antecedent conditions, canopy densities, and differences in infiltration capacities. However the proximity to establishment appeared to be the most critical time for N loss in both leachate and runoff.

Linde et al. (1998) found differences in water use and antecedent conditions between creeping bentgrass and perennial ryegrass turfgrasses that influenced the hydrology of the site under the two different plant species. In our study, the higher the soil moisture content was prior to the initiation of a storm event, the more likely we were to see significant leaching and runoff volumes from the site due to the decreased storage capacity of the soil. Morton et al. (1988) found similar differences as they observed higher N leaching under irrigated versus non irrigated Kentucky bluegrass lawns. In our study, the soil maintained the highest antecedent conditions in the winter and spring months when the plants were dormant and not actively taking-up water from the soil which led to the largest loading of leachate from all lawn types (Fig 1-2 C and D). The majority of the precipitation events that caused leachate and runoff during the 27 month period of study had precipitation rates well below the infiltration capacities of the soil. Therefore, the majority of precipitation was able to infiltrate the soil until soil saturation occurred at which point saturation excess runoff would occur. This is further supported by the finding that there were never hydrological differences among the varying degrees of slopes that were accounted for in the experimental design. Plots with higher antecedent moisture conditions reached saturation at a faster rate, resulting in gravimetrically forced drainage as leachate due to lack of storage capacity along with the saturation excess runoff.

Of all nutrients and nutrient forms analyzed in leachate during our study, NO_3^--N is the nutrient that appears to have the greatest potential to contaminate waterways through leaching and lawn type had a significant effect on nitrate leaching. A lawn consisting of primarily broadleaf weeds provided a significant reduction in N loading in leachate, and significantly reduced the loading of NH_4^+-N in runoff which was almost entirely contributed by fertilizer applications. There was 1.81% of total N lost in leachate under fertilized turfgrass conditions, but similar values were seen under other lawn types that received no supplemental fertilization (Fig 1-4). This finding provides evidence that lawns and landscapes that do not receive supplemental or cosmetic applications of fertilizers, can still contribute to nutrient loading in waterways similarly to lawns that are fertilized. However, the 100% crabgrass lawn and the 100% broadleaf lawn were the only two treatments that consistently had NO_3^--N levels <10 mg L^{-1} for the majority of the study.

These finding suggests that regardless of plant canopy composition and inputs, natural processes such as mineralization have the ability to contribute to the total loading of N in leachate and runoff. Total leachate and runoff nutrient loading was highly dependent on the total volume lost form the lawn types, and regardless of nutrient concentrations, if substantial volumes of water were transported from the soil profile, significant nutrient loading could occur. Promoting plant canopies that can intercept nutrients and deplete the soil profile of moisture content can reduce the total loading of nutrients to ground and surface water. It appears that well maintained turfgrass, (i.e., appropriately fertilized and controlled for pests) can efficiently use water and efficiently utilize applied fertilizers in the soil. Plant interception of nutrients transported through mass flow and high water use can provide a larger unsaturated soil capacity for precipitation infiltration and therefore fewer nutrients available to load into leachate.

If soil conditions are kept near saturation, the onset of a runoff or leaching event will require less precipitation to saturate the soil and subsequently generate saturation excess storm flow. In areas where adequate natural precipitation occurs for plant needs such as the Northeast of the U.S., supplemental irrigation should be avoided unless periods of drought or below average precipitation occur.

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CHAPTER 2

THE FATE OF PHOSPHORUS IN THE URBAN LANDSCAPE IN RESPONSE TO LAWN SPECIES COMPOSITION AND MANAGEMENT

Abstract

The urban landscape has become an important source of non-point pollution of fertilizers and pesticides to surface water bodies. The concern over nutrient and pesticide fate from the urban landscape questions the validity and applicability of cosmetic applications of fertilizers and pesticides to an urban lawn to improve aesthetic conditions. A field study was conducted to monitor phosphorus concentrations and mass loss in leachate and runoff from eight different lawn types and management practices. Thirty two lawn types were constructed on 13-15% slopes on an Arkport sandy loam soil (coarse-loamy, mixed, active, mesic Lamellic Hapludalf) and were designed to monitor leachate and runoff generated from naturally occurring precipitation events. Runoff and leaching events samples were analyzed for orthophosphate (PO_4^{3-}-P), total Kjeldahl phosphorus (TKP). Significant differences in leaching and runoff volumes were observed among lawn types. The application of fertilizers to lawns significantly reduced runoff volumes when no supplemental irrigation was applied. Orthophosphate leaching occurred in response to fertilizing; however, concentrations never reached levels known to induce eutrophication. Phosphorus loads in runoff related to fertilized lawn types were minimal and the percent lost in runoff was <0.5% for P. Significant levels of P where lost from all lawn types, but the largest loads in runoff occurred in crabgrass and unfertilized turfgrass lawns. The application of supplemental fertilization appears to have the ability to reduce runoff volumes through increased density compared to some non-fertilized lawn types. The percent fertilizer lost through runoff was nearly an insignificant amount of total fertilizer applied for the 27 month study and does not

appear to pose a significant environmental risk at considerable slopes of 13-15%.

In recent years, concern over water quality has brought closer scrutiny upon the use of fertilizers and pesticides in the landscape and the impact they may have. For many, the use of fertilizers and pesticides in the landscape appear to serve no greater function than providing a cosmetic benefit that improves the aesthetic appearance and financial value of the lawn. The application of nutrients and pesticides to turfgrasses can have several fates, but the loss to waterways is a common path. Research studying the dynamics of nutrient loss in agricultural and turfgrass systems has shown that nutrients may exit off-site through both above ground surface runoff and underground leaching.

Phosphorus is considered to be the most limiting nutrient in lakes for algal growth and is usually a larger concern in freshwater systems (Corell, 1998).Eutrophication can occur with dissolved reactive P concentrations of 0.01 mg L^{-1} and total phosphorus concentrations of 0.02 mg L^{-1} (Sharpley and Rekolainen, 1997). Soluble P is bioavailable for algal uptake and is the focus of the majority of research pertaining to phosphorus. For most lakes, streams, reservoirs, and estuaries, concentrations of 100 μg total P L^{-1} are unacceptably high and concentrations of 20 μg L^{-1} are can cause eutrophication (Corell, 1998). Phosphorus may be chemically or enzymatically hydrolyzed to orthophosphate, which is the only form of P that can be assimilated by bacteria, algae and plants (Corell, 1998; Mathers et al. 2007). The plant available form of P is pH dependent with $H_2PO_4^-$ being dominant at low pH (< 7.2) and HPO_4^{-2} dominates at a higher a pH (> 7.2) (Havlin et al., 2005). Phosphorus is considered to be immobile in soil, and residual P availability can persist for years dependent on soil type, and vegetative removal.

Phosphorus can be transported to waterways either through soluble or particulate forms bound to sediment (Carpenter et al., 1998; Sharpley et al., 2000).

The amount of P lost to surface waters tends to increase with the P content of the soil (Soldat and Petrovic, 2008; Carpenter et al., 1998). Dissolved nutrients are generally more damaging to the environment than particulate forms because they can be transported further into waterways and tend to be more bioavailable (Mathers et al., 2007). Sharpley et al. (1992) found the percent bioavailability of particulate P transported in runoff from watersheds decreased with an increase in sediment concentrations of runoff. However, one should not assume that particulate forms of P or dissolved organic P are inert in waterways because under appropriate conditions, these forms can be converted to dissolved orthophosphate (Corell, 1998).

Knowing that fine particles are the major source of waterway pollution, grass is very effective for sediment control (Deletic, 2005). Due to its dense canopy, turfgrass is often used as an erosion control method and can reduce erosion and runoff significantly (Krenitsky et al., 1998) which could reduce a long term bioavailable source of P associated with particulates (Sharpley et al., 1992).

With few exceptions, surfaces waters receive most of their P in surface flows rather than in groundwater. Total P losses in leachate from turfgrass have shown to be to be low and the majority of P loss is seen in runoff (Barton et al., 2006). The mass of P transported to waterways from turf is directly correlated to the rate of application (Shuman, 2004). In turfgrass and prairie plots soluble P comprises the majority of P in runoff due to relatively dense vegetative cover in buffer strips which inhibits soil erosion and corresponding particulate phosphorus (Steinke et al., 2007). In grasslands, Watson et al. (2007) found a highly significant ($p < 0.001$) increase in annual Olsen-P status in plots receiving P fertilizer, which increased with the rate applied. Regardless of fertilizer source, the highest nutrient concentrations in surface flow from turfgrass tend to occur in the first runoff event following application of nutrients (Shuman, 2002; Easton and Petrovic, 2004). Ultimately, even low maintenance landscapes can be a source of P in urban environments due to natural and necessary P uptake by leaves and subsequent leaching from vegetation (Steinke et al., 2007).

In most cases, surface waters receive most of their P through surface flows rather than through groundwater since phosphorus binds strongly to most soil and sediment (Corell, 1998) therefore the focus of nutrient losses in runoff tend to be on P while N losses are more strongly associated with leaching loss. Fertilization of the turfgrass can create higher shoot density, higher soil infiltration rates, lower soil moisture levels and subsequently less runoff and nutrient loss (Easton and Petrovic, 2004). Lowering mowing height reduced runoff losses from buffers mowed at 1.3 cm compared with 3.8 cm due to increased density in Bermudagrass and reduced losses of all pesticides, NH_4^+-N, and PO_4^--P in runoff water compared to plots with no buffers (Cole et al., 1997). Steinke et al. (2007) found that dense turfgrass plots reduced runoff losses by 54% compared to prairie plots in 2004 and that total P losses from prairie were four times greater than turf during that same growing season.

Turfgrass cover has been shown to reduce runoff volume and the transport of NO_3-N and P through a reduction of soil moisture prior to rainfall events and that rainfall-runoff dynamics were closely related to initial moisture conditions (Shuman, 2004). Runoff events typically occur in response to intense rainstorms, and runoff volumes are greater when the antecedent soil moisture content is higher (Linde et al., 1995; Srinivasan et al., 2007). Winter can be an important time for runoff losses, especially if the soil freezes and forms an impermeable surface for precipitation and snowmelt. Easton and Petrovic (2004) found in the second year of a two year study that snowmelt or runoff on frozen soils accounted for 62% of total runoff collected over the entire year for Kentucky bluegrass and perennial ryegrass plots. Steinke et al. (2007) found a direct relationship between runoff volume and season and on an annual basis, the effect of vegetation type was negligible over 80% of the annual runoff occurred during the winter when the ground was frozen. Seasonal variation may not only affect runoff volumes, but may affect nutrient and chemical loads in runoff, therefore, it is important to have a full understanding of potential risks associated with runoff losses for all times of the year and not just when the plants are actively growing.

In turfgrass, P deficiency symptoms or rarely observed in part due to turfgrasses fibrous root system that is very effective at mining and taking up available P from the soil. Many turfgrass management programs have begun to promote lower P application rates to reduce the residual amounts of P found in the soil that can potentially have an environmental impact to waterways if lost through surface or subsurface flow

The objective of this study was to examine the effect of perceived cosmetic applications of nutrients and irrigation to newly established turfgrass lawn has on P mobility in surface and ground water, and to monitor the effects these applications have on the density and composition of the species within the plots. These lawns were compared to lawns composed of turfgrass, annual and perennial weedy species that receive no maintenance beyond mowing for loading of P species to surface and groundwater.

Materials and Methods

The following study was conducted at the Turfgrass and Landscape Research Center at Cornell University in Ithaca, NY. A southwest facing hill-slope having a slope range of 13-15% and measuring 39 m by 6.1 m was stripped of all sod to a depth of 3.8 cm. Surface soil was removed with a bulldozer and stockpiled for later use. The subsoil was removed to a depth of 45 cm and the underground subsoil was thoroughly compacted with the bulldozer by driving over the site repeatedly. The remaining subsoil and topsoil was replaced and compacted in 15 cm lifts with the bulldozer to ensure consistent compaction levels and to provide conditions similar to high soil disturbance of a new home construction site.

The hill slope was subdivided into thirty two 1.2 m wide x 6.1 m long plots. Lysimeters were constructed similar to Miltner et al. (1996) made of polyvinyl chloride (PVC) pipe with a 61.0 cm diameter and 91.4 cm length. Each length of PVC was driven into the ground vertically in the top 1.2 m section of the hill slope, using a loader until the top of the lysimeters was flush with the surface of the soil. All attempts were made to remove the lysimeters with the soil core intact, and once removed, a 10 cm gravel layer was placed at the bottom, and a flat sheet of 12.7 mm thick PVC with a diameter of 61.0 cm was attached to the base with PVC cement to ensure a tight seal. If a soil core was not removed intact, the soil was removed in 15 cm lifts and placed in the lysimeters in the same sequence it was removed, and then tamped to ensure the same quantity of soil fit into the lysimeters to maintain the bulk density of the soil profile. Approximately one half or two replicates of the lysimeters were removed with the core intact and the other half were removed by hand. Once all bases were affixed to the lysimeters, they were lowered back into the excavated site they were removed from.

A trench 1.0 m deep was placed on the downhill slope of the lysimeters to give access to the base and a 12.7 mm diameter exit drain was placed at the bottom of each lysimeters. Polyethylene pipe with a diameter of 12.7 mm was connected to each exit drain with a barbed fitting and tubing from eight lysimeters was run downhill to a common collection area for leachate sampling in 4000 mL plastic containers. After the installation of the lysimeters, trenches were backfilled and compacted to prevent settling. The entire area was covered in approximately 7-8 cm of topsoil and hand raked in preparation for establishments.

The study consisted of a randomized complete block design with the variation in the slope across the plots used as the blocking factor to provide four blocks with eight treatments per block (Table 2-1). Treatments consisted of Kentucky bluegrass (*Poa pratensis* L.) (KB) lawn that received applications of fertilizers (F) and pesticides (P) using consumer lawn products (The Scotts Comp., Marysville, OH) that provide 4 fertilizer applications and 4 pesticide applications; pendimethalin [N-(1-ethylpropyl)-3,4-dimethyl-2, 6-dinitrobenzenamine] in spring, mecoprop [2-(4-Chloro-2-methylphenoxy)propionic acid] + 2, 4-D (2,4-dichlorophenoxyacetic acid) in late spring and early fall, and bifenthrin [(2-methyl-1,1-biphenyl-3-y1)-methyl-3-(2-chloro-3,3,3-trifluoro-1-propenyl)-2,2-dimethyl cyclopropanecarboxylate] in summer, and supplemental irrigation (I). Irrigation was applied when soil moisture content decreased below 20% relative water content which was determined to be the level of 50% depletion of plant available water (Brady and Weil, 1999). Plots were irrigated with 125 mm of water at each irrigation event with a below ground irrigation system installed in the four KB plots that receive supplemental irrigation; irrigating at a rate of 75 mm hr^{-1}. The volume of water applied at each irrigation event did not induce runoff. The second treatment was the same as the first described above, but without supplemental irrigation. The third treatment consists of KB receiving all four fertilizer applications, but no pesticide and no supplemental irrigation applications. The fourth treatment consists of KB with no applications of pesticides, fertilizers, or supplemental irrigation and was treated as the untreated control plot. The remaining four treatments represent more common weedy lawns found in a humid temperate landscape that receive no fertilizers, pesticides, or irrigation. The fifth treatment was a lawn consisting primarily of typical cool season broadleaf weeds that is only mowed and will be referred to a 100% broadleaf weed lawn. The sixth treatment was a lawn with 50% KB lawn plugged with 50% broadleaf weeds with plots only mowed. The seventh treatment was established entirely as annual grassy weeds, and was seeded with smooth crabgrass (*Digitaria ischaemum*.) and will be referred to as a 100% crabgrass lawn. The eighth plot consists of mixture of 50% KB and was 50% smooth crabgrass that will only be mowed. All plots were mowed weekly at a height of 5 cm with a rotary mower similar to those used in a home lawn and all clippings were removed.

Plots that contain KB were sodded on 3 July 2007 with KB containing four different cultivars (Odyssey, Liberator, Rugby II, and Total Eclipse). Plots that consist of both 50% broadleaf weeds and 50% crabgrass weeds, had 108 mm diameter plugs of KB removed and plugs of broadleaf plantain (*Plantago major*), white clover (*Trifolium repens*), and common dandelion (*Taraxacum officinale*) were plugged into place while plugs of smooth crabgrass were used on the 50% crabgrass plots. All plots were plugged with approximately four hundred 108 mm diameter plugs until visually a 1:1 ratio of weeds to KB was obtained. For plots consisting of primarily broadleaf weeds, sod with approximately 90-100% of the described broadleaf weeds was stripped from a site at the research center and placed on the plots. The 100% annual grassy weed plots were seeded with smooth crabgrass at a rate of 614 kg ha^{-1} and over seeded yearly at a rate of 200 kg ha^{-1} on 15 May 2008 and 12 May 2009. All plots were irrigated to facilitate root establishment of sodded plots and seedling establishment of crabgrass plots for four weeks. Plastic landscape edging was placed between each plot to a depth of 50 mm to confine flow of surface water within plots.

Table 2-1. Application dates and product descriptions of the fertilizer and pesticide combinations applied to the Kentucky bluegrass turfgrass plots.

Application date	Product N-P-K ratio	Nitrogen applied	Phosphorus applied	Potassium applied	Active ingredient
			——kg ha^{-1}——		
3 September 2007	28-3-8	36.5	1.7	8.7	Bifenthrin
24 September 2007	26-3-12	30.1	1.5	11.5	2,4-D; Mecoprop
23 April 2008	30-3-4	41.2	1.8	4.6	Pendimethalin
26 May 2008	28-3-3	39.2	1.8	3.5	2,4-D; Mecoprop
28 July 2008	28-3-8	36.5	1.7	8.7	Bifenthrin
3 October 2008	26-3-12	30.1	1.5	11.5	2,4-D; Mecoprop
29 April 2009	30-3-4	41.2	1.8	4.7	Pendimethalin
26 May 2009	28-3-3	39.2	1.8	3.5	2,4-D; Mecoprop
22 July 2009	28-3-8	36.5	1.7	8.7	Bifenthrin
16 September 2009	26-3-12	30.1	1.5	11.5	2,4-D; Mecoprop
Total applied		**360.6**	**16.8**	**76.9**	

During establishment, a 155 mm wide by 200 mm deep trench was placed at the bottom of all plots running perpendicular to the hill slope. A 127 mm wide, 1.20 m long channel drain gutter was cut to the length of each plot and placed with a 0.5% slope at the base of each plot with a 12.7 mm diameter drain placed at one end connected to polyethylene pipe that led the water to a collection site for the surface runoff water. All containment gutters were covered with 3.2 mm thick metal sheeting to stop rain and outside contaminants from entering the collection site. The drains for the channel gutters ran downhill where piping was grouped for every two plots collection area where soil was excavated and a 95 L plastic storage container was placed in the soil. The polyethylene pipe entered through the uphill side of the container and was extended over 100 mL tipping buckets. Tipping buckets were leveled and placed on 12.7 mm thick plywood with holes drilled large enough for 1000 mL sampling bottles to be placed below the buckets. Tipping buckets were connected to event data loggers (Onset Computer Corp., Pocasset, MA, model HOBO® pendent event logger) and every tip was calibrated prior to use by pouring 1000 mL of water into each collection gutter periodically and measuring the number of tips for the given volume. The number of tips and frequency of tips were measured for each storm event and logged to equate total runoff losses from each plot. Runoff storm events were monitored and classified as a precipitation event when runoff occurred to any plot within the study.

Subsamples of leaching and runoff and samples were taken within 6 h of precipitation events, and frozen immediately until further analysis. Subsamples were compiled into sampling periods or bulking based on the frequency of storms in a time period to obtain 3-6 precipitation events that caused runoff per period and to reduce the number of samples to analyze. Subsamples were bulked proportionally to the volume of leachate or runoff for the given storm event to create bulked subsamples with nutrient concentrations proportionate to the actual storm events. Once bulked, subsamples were filtered through a 45 µm filter to remove sediment and to analyze nutrients.

Precipitation events that produced leachate were separated by rainfall; however, often multiple storms occurred within several hours to days of each other and leachate volumes could not be separated from events. In such a case, total leachate volumes were recorded for the entire precipitation event and the event was labeled as a single precipitation event with the precipitation amounts recorded for the entire period. Collection of leachate occurred in 4000 mL containment bottles in which the total volume was measured volumetrically and recorded at collection.

Samples analyzed for dissolved reactive phosphorus (PO_4^{-3}) were reacted with acidic molybdate in the presence of antimony to form an antimony phospho-molybdate complex which is reduced by ascorbic acid to an intensely blue complex that was measured colorimetrically at 660 nm (EPA Method 128-A). Total filtered Kjeldahl phosphorus (TKP) samples were digested according to (EPA method 136-A) under the presence of sulfuric acid, potassium sulfate and copper (II) sulfate to convert organic compounds to orthophosphate. Samples were then analyzed according to the protocols for the given compound described above in the Turfgrass Research Laboratory at Cornell University Ithaca, NY 14853 (Seal-Analytical, Mequon, WI, Model: AQ2).

Plant density was measured monthly through digital photo analysis. A digital camera (Olympus Inc., Model: 5.0 Megapixel Stylus) was mounted on a monopod that stood 1.0 m above the plot and extended out 0.60 m horizontally away from the vertical axis to allow the camera to be positioned directly above the plots without obstruction (Richardson et al., 2001) and photos of the top, middle, and bottom portions of the plots are taken. All photos are taken between the 1100 and 1300 h. to optimize light and provide consistent conditions. Photos were analyzed using digital imaging software (SPSS, Inc., Chicago, IL, Sigmascan Pro v. 5.0) by separating hues related to plant color of 45 to 107 and a saturation range from 0 to 100. Hue ranges between 57 and 107 have been shown to selectively identify green leaves in an image (Richardson et al., 2001), but through photo analysis, a wider range was found to be more optimum to isolate plant color for this study. Total pixels related to green leaves were then divided by the total pixels in the image to give a percent coverage of the plots.

Volumetric moisture content of plots were taken periodically using time domain reflectometry (TDR) through a water content probe (Spectrum Technologies Inc., Plainfield, IL, model: FieldScout TDR 300) with 120 mm rods to monitor irrigation frequency and antecedent conditions. Measurements were taken at the top, middle, and bottom of each plot to quantify dynamics that may affect water movement above or below the soil surface.

The study was initiated on 10 August, 2007 when the first supplemental irrigation event was applied to KB plots receiving irrigation plus pesticides and fertilizer applications and storm events were monitored for 27 months through 1 November, 2009. During the second winter of the study, frost tubes (McCool and Molnau, 1984) were place in the center of each plot to determine the relationships between cover type and soil freezing, and also the relationship between soil freezing depth and runoff and leaching frequencies. Data was monitored primarily during the initial soil freezing period or when climatic conditions were favorable for snowmelt or rainfall that may induce leaching.

On 2 September 2008, 7.3 cm diameter undisturbed soil cores were taken 6 cm deep from each plot. A soil core was taken from the top, middle, and bottom portion of each plot and stored in a cooler at 4°C. Samples were analyzed for distribution of pore space, penetrometer resistance, and bulk density as described in (Moebius et al, 2007). Soil cores measuring 2.2 cm diameter x 8 cm deep were extracted on 26 September 2007, 21 October 2008, 26 June 2009 and 12 August 2009 from several random locations within each plot and homogenized to create a soil sample representative of each plot. Samples were used to determine soil particle distribution through the hydrometer method (Sheldrick and Wang, 1993), Morgan extractable phosphorus (Morgan, 1941), soil pH in 0.01 M $CaCl_2$ (Hendershot et al, 1993): organic matter from loss on ignition (Karam, 1993).

Pre-established soil infiltration rates were measured within each block on 23 May 2007 using a miniature rain infiltrometer (Ogden et al. 1997) and secondary infiltrations measurements were taken on 5 June 2008 and 23 September 2009 within each plot to monitor changes in infiltration through the establishment period.

Statistical analysis

All runoff nutrient data was subjected to analysis of variance using a mixed model in JMP 8.0 (SAS institute, 2009) with repeated measures. Treatment and sample period were treated as fixed effects, while the blocking factor was treated as a random variable in the model. When significant differences were observed, means were separated using Tukey's HSD test with $\alpha = 0.05$ probability level.

Soil sample and plot characteristics were subjected to ANOVA with repeated measures with the exception of undisturbed soil core data that was taken only once. If following ANOVA, there was no significant effect of time on a measurement, data for the plot was averaged over time and means were separated using Tukey's HSD test with $\alpha = 0.05$ probability level when significant effects were observed through ANOVA.

Results and Discussion

Hydrology

Based on the 27 month duration of the study, there were only five storm events that produced >25 mm or rainfall in a 24 hour period (Fig. 2-1). This factor becomes important, because based on two separate measurement dates the soil infiltration capacities of all the lawns were >100 mm hr^{-1} (Table 2-2). In fact, during the course of the study, we never observed a storm event that produced a precipitation rate large enough to force infiltration excess flow or Hortonian flow (Horton, 1933; Horton, 1940) from the plots. Therefore, it can be ascertained that the runoff sampled and measured in this study was derived primarily from saturation excess flow conditions (Hewlett and Hibbert, 1963) and similar finding have been seen for the region by Walter et al. (2003).

In the 27 month period of our study, there were 68 leachate inducing precipitation events and 74 runoff inducing precipitation events (Table 2-3) observed which were combined to give 16 sample periods where subsamples were taken from for analysis. There were important differences observed and measured among the treatments for factors known to impact the rate and volume of leachate and runoff that can be lost from a site (Tables 2-2). Lawn types were found to affect soil volumetric water content (VWC), soil infiltration capacity (IC), vegetative density, soil organic matter (SOM), and the frost depth (Table 2-2).

One of the most relevant measurements in predicting runoff is the antecedent moisture conditions (Linde et al., 1998; Srinivasan et al., 2007). Although not always taken directly prior to the onset of a leaching or runoff event, the differences in VWC among the lawn types was quite obvious (Table 2-2) and can be related to several factors including; plant water use, IC, vegetative density, and soil water storage capacity. Among the treatments, the irrigated turfgrass plots consistently had significantly higher soil VWC due to the applied supplemental irrigation (Table 2-2). The lawns containing portions of weedy species and the unfertilized turfgrass plots had significantly lower VWC than the irrigated turf, however, those plots consistently had higher VWC than the two turfgrass plots that were fertilized and did not receive supplemental irrigation and the 100% crabgrass plot. The differences observed among the non-irrigated lawns are likely attributed to differences in either water holding capacities and drainage rates of the soil or of the differences in evapotranspiration (ET) among the canopies, because all plots received the same amount of precipitation aside from the irrigated turfgrass plots. This suggests that fertilized turfgrass has a higher ET or plant water use than the other lawns containing combinations of the weedy species. Fry and Butler (1989) found ET rates of common lawn weeds, with the exception of white clover to be less than 'Merion' KB. One exception to that theory is the fact that the 100% crabgrass pots had similar soil VWC to the fertilized turfgrass plots. However, the crabgrass plots had significantly higher runoff volumes than the fertilized turfgrass plots (Fig. 2-2) with lower VWC and differences are most likely caused by exposed bare soil contributing to surface sealing of the soil, therefore reducing the amount of precipitation that infiltrated into the soil.(Gross et al., 1991; Carroll et al., 2000)

Vegetative density is an important factor in runoff dynamics and has been shown that high density reduces runoff volumes and frequencies from turfgrass (Linde et al., 1995; Easton and Petrovic, 2004). The differences in VWC we observed among the fertilized and unfertilized turfgrass treatments and the most prominent finding to support the differences observed in VWC in unfertilized turfgrass was the lower percentage of vegetative cover compared to the fertilized plots (Table 2-2; Fig. 2-2). The reduction in canopy area or density would directly reduce the plant transpiration and water use and allow larger volumes of water to be stored in the soil, if infiltration was not restricted through surface sealing. The unfertilized turfgrass and the 100% crabgrass lawns at times had significantly lower vegetative density values than any other plot, along with having the highest runoff volumes similar to the irrigated turfgrass plots (Fig 2-3 C and D).

The amount of soil organic matter was related to lawn type. Lawns that had the lowest SOM content (Table 2-2) also had the lowest vegetative density (Table 2-2), suggesting that less plant material was available for natural recycling and incorporation into the soil surface as an organic matter component.

Leachate hydrology

There were differences observed in leaching volumes from the different treatments, and the majority of loading occurred within the first six sample periods of the study (Fig. 2-3 C and D). This period contributed significantly to the overall load of leachate for the plots due to climactic conditions that kept the soil from freezing very deeply. We observed several freeze thaw cycles during that period that led to large amounts of snowmelt and rainfall over saturated soil conditions. From the initial establishment period, the slope from each lawn type decreased and the remaining leachate inducing precipitation events contributed much less to the total leachate load from the plots (Fig 2-3 C and D).

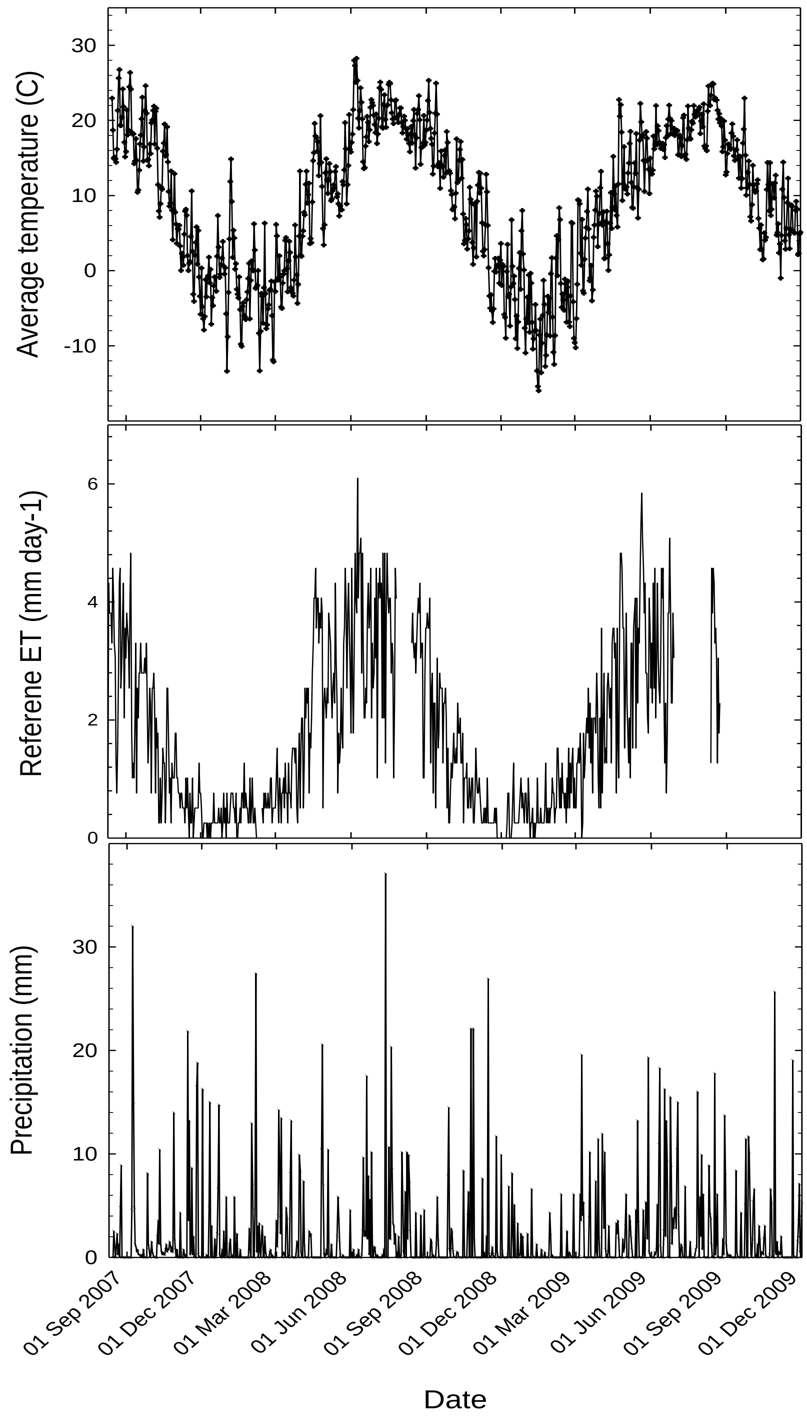

Figure 2-1. Climactic data for Cornell's Turfgrass and Landscape Research Center in Ithaca, New York for the 27 month study period.

Table 2-2. Soil and plot characteristics observed in eight different lawn types. Bulk density and soil organic matter were averaged from four different sampling dates. Infiltration rates were averaged over two different time points in two separate growing seasons using a sprinkler infiltrometer. Frost depth is the average observed frost depth over the 2008-2009 winter season.

	Bulk density	**Average seasonal frost depth**	**Infiltration capacity**	**Average soil VWC**	**Soil organic matter**
Treatment	**g cm^{-3}**	**mm**	**mm hr^{-1}**	**m^{3}m^{-3}**	**g kg^{-1}**
Broad + Turfgrass	1.42 ab	70 a	213.2 a	0.236 b†	53.3 abc
Broadleaf	1.40 b	68 ab	219.7 a	0.236 b	52.6 bc
Crabgrass + Turfgrass	1.35 b	68 ab	174.3 abc	0.237 b	53.9 ab
Crabgrass	1.52 a	69 a	112.5 bc	0.224 cd	45.6 d
Turfgrass	1.42 ab	67 ab	158.1 abc	0.234 b	49.2 cd
Turfgrass + F	1.38 b	50 bc	186.6 ab	0.225 c	53.2 abc
Turfgrass + F + P	1.41 ab	47 c	101.2 c	0.219 d	57.3 a
Turfgrass + F + P + I	1.45 ab	43 bc	116.4 bc	0.244 a	52.5 bc

†= values followed by the same letter within columns are not significantly different according to Student's t test of least significant difference ($p < 0.05$), NS = not significant

The turfgrass lawns that received fertilizer applications, but no irrigation had lower leachate volumes than the irrigated turfgrass and the non-fertilized turfgrass. Through multiple regression analysis, there were significant effects of VWC ($p=0.019$) of the plots as well as plant density ($p=0.018$) on the ratio of precipitation observed as leachate from the plots (data not shown). The overall differences observed among leaching volumes within treatments (Fig. 2-3 C and D) are representative of overall differences observed in the moisture contents of the plots (Table 2-2), as higher antecedent conditions or lower plant densities (Fig. 2-2) provided significantly higher leachate ratios to be observed from the plots.

Phosphorus leachate losses

The concentrations of P in leachate were significantly higher under turfgrass lawn types compared to the weedy lawn types, but differences were also dependent on the season samples were taken. The largest concentrations observed for both PO_4^{-3} and TKP in leachate came in sampling period 7 and 12 (Figures 2-3 and 2-4). Period 7 was the 5 April to 17 May of 2008, while the period 12 was 6 Jan to 30 March of 2009 (Table 2-3). Concentrations observed in leachate under turfgrass conditions were often measured above 0.02 mg L^{-1} (Fig. 2-3E). A level of P >0.02 mg L^{-1} is known to contribute to eutrophication of surface water , thus important in areas where groundwater contributes to base flow for streams and rivers.

Table 2-3. Leaching and runoff events and corresponding date ranges assigned to sample periods

Sample period	Date range	Leachate events	Date range	Runoff events
1	8 Sept 2007-15 Sept 2007	1-3	8 Sept 2007-15 Sept 2007	1-4
2	16 Sept 2007-27 Oct 2007	4-9	16 Sept 2007-27 Oct 2007	5-13
3	28 Oct 2007- 3 Dec 2007	10-16	28 Oct 2007- 3 Dec 2007	14-18
4	4 Dec 2007-11 Jan 2008	17-23	4 Dec 2007-11 Jan 2008	19-21
5	12 Jan 2008- 29 Feb 2008	24-30	12 Jan 2008- 29 Feb 2008	22-28
6	1 Mar 2008- 4 Apr 2008	31-37	1 Mar 2008- 4 April 2008	29-33
7	5 Apr 2008- 17 May 2008	38-39	5 Apr 2008- 17 May 2008	34-35
8	18 May 2008- 15 July 2008	40-41	18 May 2008- 15 July 2008	36-40
9	16 July 2008- 10 Aug 2008	42-44	16 July 2008- 10 Aug 2008	41-45
10	11 Aug 2008- 15 Nov 2008	45-47	11 Aug 2008- 15 Nov2008	46-50
11	16 Nov 2008- 5 Jan 2009	48-51	16 Nov 2008- 5 Jan 2009	51-53
12	6 Jan 2009- 30 Mar 2009	52-55	6 Jan 2009- 30 Mar 2009	54-57
13	31 Mar 2009- 6 May 2009	56-57	31 Mar 2009- 18 June 2009	58-62
14	7 May 2009- 30 June 2009	58-61	19 June 2009-26 July 2009	63-66
15	1 July 2009- 29 Aug 2009	62-64	27 July 2009- 29 Aug 2009	67-72
16	30 Aug 2009- 15 Nov 2009	65-68	30 Aug 2009- 15 Nov 2009	73-75

Total loading of P in leachate was minimal for the weedy turfgrass plots; however, there was a trend towards higher losses as the plots matured (Figure 2-4 H). The unfertilized turfgrass contributed the most significantly to total loading of P in leachate among all lawn types, while differences in other lawn types were not as pronounced (Fig. 2-4 G and H). Phosphorus loads observed in leachate varied with time due to the same interaction among lawn type, sample period, and year (Table 2-4) that affected leaching losses and we can ascertain that losses were highly dependent on leaching volumes. Generally, the load of P in leachate was small and P loading cannot be contributed to fertilizer because the non-fertilized turfgrass plot had the highest cumulative loading of PO_4^{-3} and TKP in leachate than any other treatment in the study (Fig 2-3 G and H).

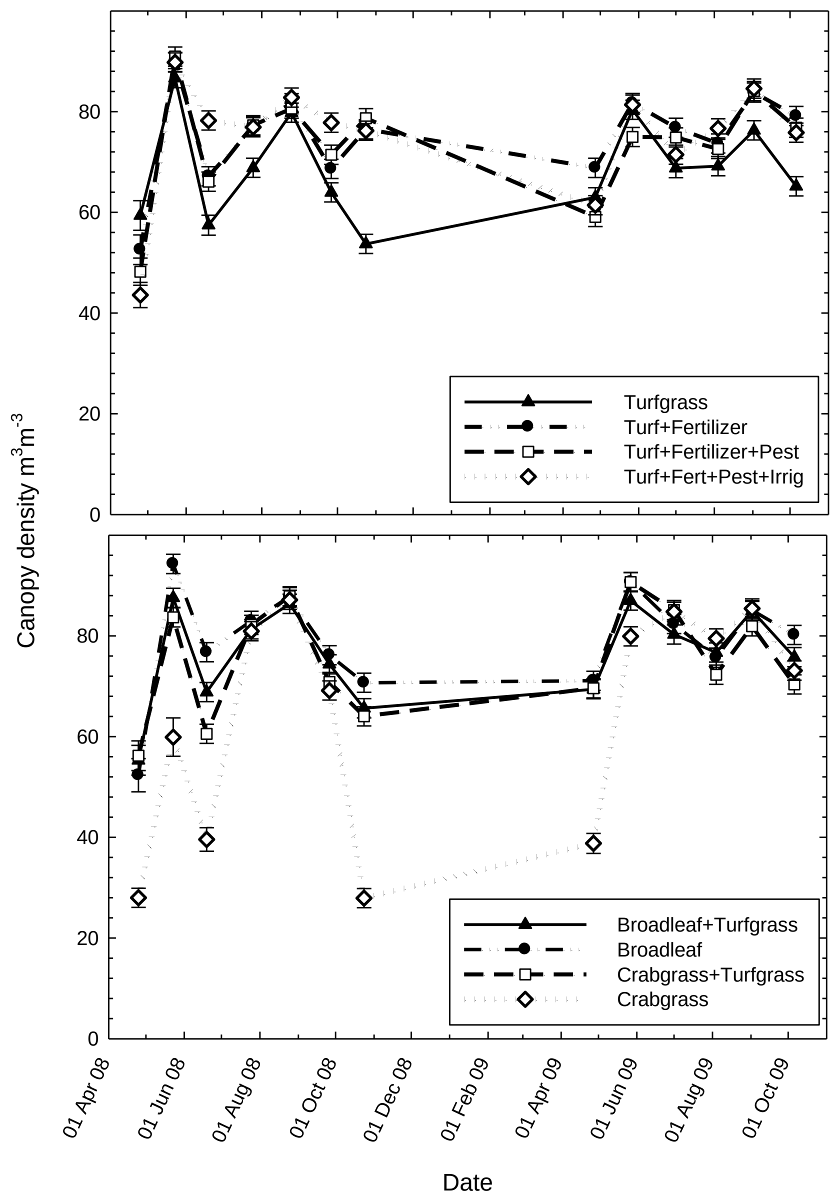

Figure 2-2. Plant densities measured through digital image analysis over two growing seasons for eight different lawn types

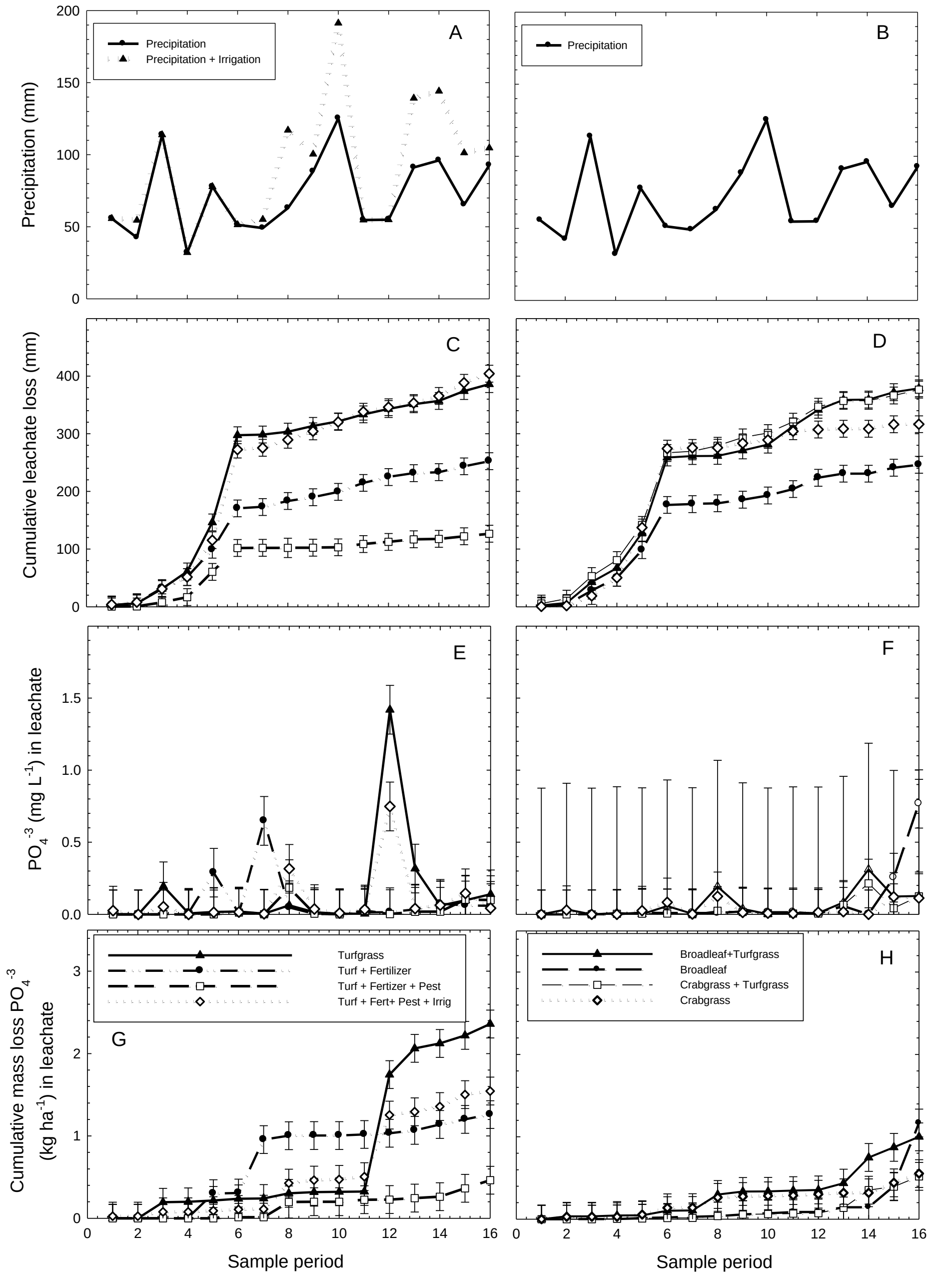

Figure 2-3. Precipitation and irrigation inputs (A)(B), cumulative leachate loss (C)(D), PO_4^{-3} concentrations (E)(F), and cumulative leaching of PO_4^{-3} (G)(H) measured from eight different lawn types through natural precipitation events over 16 sample periods over 27 months.

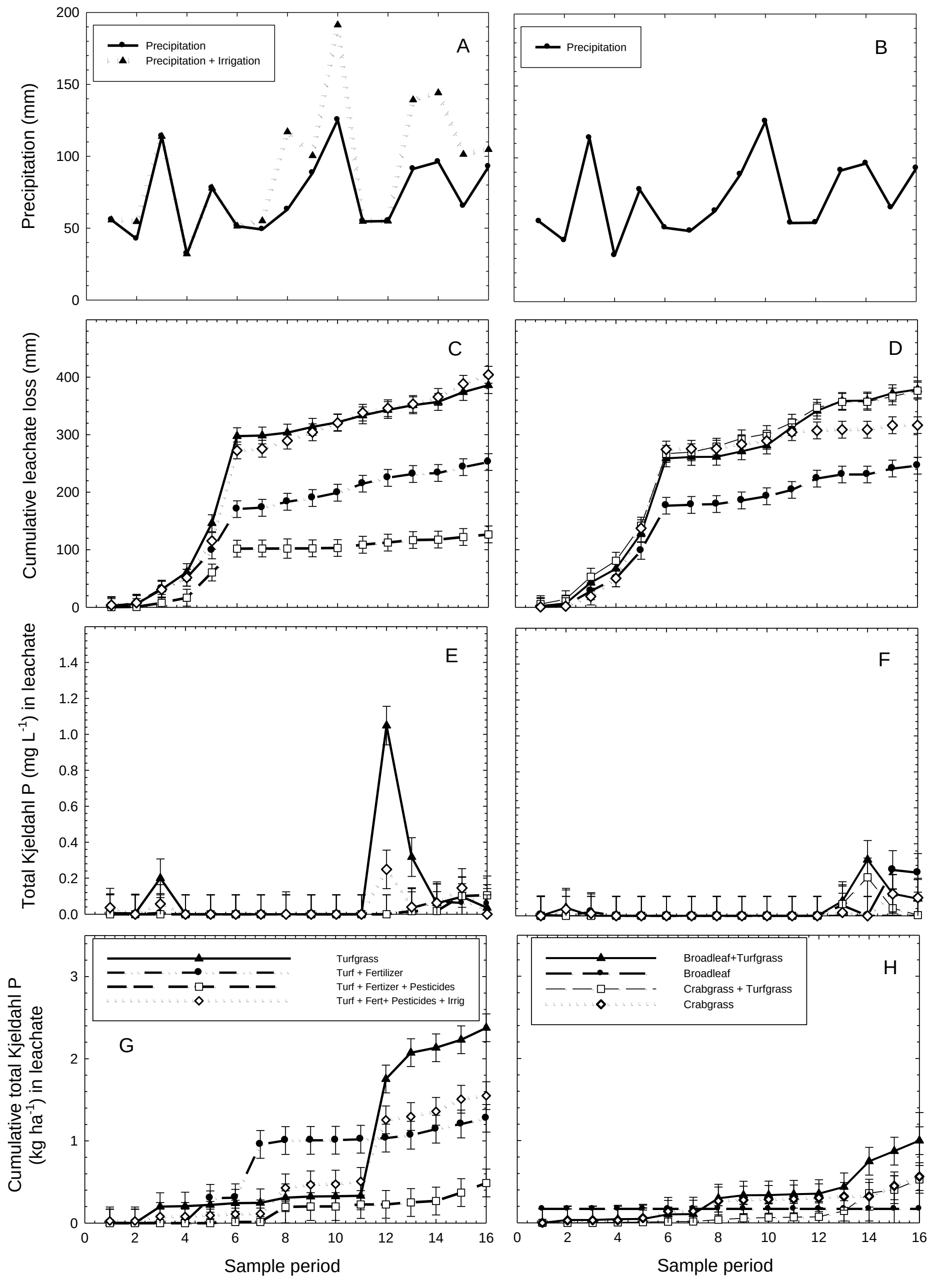

Figure 2-4. Precipitation and irrigation inputs (A)(B), cumulative leachate loss (C)(D), total Kjeldahl P concentrations (E)(F), and cumulative leaching of total Kjeldahl P (G)(H) measured from eight different lawn types through natural precipitation events over 16 sample periods over 27 months.

Table 2-4. ANOVA table for P concentrations (mg L^{-1}) and mass loss (kg ha^{-1}) of P measured in lawn leachate. ANOVA was performed by multiple regression analysis using a generalized linear model.

		PO_4^{-3}	**Total P**	**PO_4^{-3}**	**Total P**
Source of variation	***df***	***concentration***		***Mass loss***	
		p-value			
Treatment	7	ns	ns	<0.001	<0.001
Sample period	1	ns	ns	ns	ns
Season	3	ns	ns	ns	ns
Year	1	ns	ns	ns	ns
Treatment*Period	7	ns	ns	ns	ns
Treatment*Season	21	ns	ns	ns	ns
Treatment*Year	7	ns	ns	0.037	ns
Sample period*Year	1	0.037	0.05	0.006	ns
Season*Year	3	ns	0.031	ns	ns
Treatment*Period*Year	7	ns	ns	<0.001	<0.001

† Indicates non-significant differences at α=0.05 level.

Runoff hydrology

In our study, there were significant differences observed between the lawn types on the cumulative loading of runoff from the plots (Table 2-5). Runoff volumes for the 27 month period revealed important differences among treatments based on canopy characteristics (Fig. 2-5 C and D). The fertilized turfgrass plots that did not receive supplemental irrigation had significantly lower portions of runoff leave the canopy compared to the 100% crabgrass and the turfgrass plots that received supplemental irrigation (Fig. 2-5 C and D). There was a strong trend for higher runoff volumes over-time from the unfertilized turfgrass lawn and the 100% crabgrass as there was a reduction in canopy density. Plots that that had greater canopy density that promoted higher water use provided a convoluted canopy to reduce runoff velocity as seen in other studies (Linde et al., 1995; and Easton and Petrovic 2004). However, over-time the canopy density equalized as weedy species and began to fill voids left in the canopy and differences in canopy densities were no longer different (Fig 2-2).

There were no different plot observations made between the turfgrass lawn type characteristics that receive supplemental irrigation and those that do not with the exception of the soil VWC (Table 2-2). The higher antecedent conditions observed for the irrigated turfgrass significantly contributed to higher runoff volumes observed from the plots ($p=0.026$) as seen from a multivariate analysis. On the other hand, the 100% crabgrass plots had similar soil VWC's to the fertilized turfgrass plots, but had significantly lower vegetative density measurement (Table 2-2).

Soil VWC differences along with the difference in soil freezing depth (Table 2-2), were the most important differences observed between the treatments and help explain the differences seen in the volume of runoff lost from the plots. The turfgrass plots reduced the total time in the season that the ground was frozen compared to the other lawn types, which provided the potential for precipitation to infiltrate into the soil whereas the other plots had an impermeable surface that would force surface runoff over frozen ground.

Phosphorus concentrations in runoff

The effect of lawn cover and treatments had on nutrient concentration varied dependent on and nutrient form. There were significant two way interactions observed for the two forms of P measured in runoff and the interaction of season and study year was the most significant factor in P loss in runoff (Table 2-5). After thorough analysis, the difference observed for PO_4^{-3} concentrations in runoff were primarily caused by the concentrations observed in the turfgrass+fertilizer+pesticide treatment, which were significantly higher than the two crabgrass treatments, the turfgrass and the turfgrass+fertilizer+pesticides+irrigation treatments (Fig 2-5 E and F). However, the highest treatment concentrations were not significantly different from the broadleaf treatment plots or the turfgrass+fertilizer plots (Fig 2-5 E and F).

Table 2-5 ANOVA table for P concentrations (mg L^{-1}) and mass loss (kg ha^{-1}) of P measured in lawn runoff. ANOVA was performed by multiple regression analysis using a generalized linear model.

		PO_4^{-3}	**Total P**	**PO_4^{-3}**	**Total P**
Source of variation	***df***	***concentration***		***Mass loss***	
		p-value			
Treatment	7	<0.001	<0.001	<0.001	<0.001
Sample period	1	ns	ns	0.039	ns
Season	3	0.004	<0.001	ns	ns
Year	1	0.014	ns	ns	ns
Treatment*Sample period	7	ns	ns	ns	ns
Treatment*Season	21	ns	0.032	ns	ns
Treatment*Year	7	0.040	0.043	ns	ns
Sample period*Year	1	0.002	ns	ns	ns
Season*Year	3	<0.001	<0.001	ns	ns
Treatment*Period*Year	7	ns	ns	ns	ns

† Indicates non-significant differences at the α=0.05 level

The effect of sample period and season was apparent for PO_4^{-3} (Fig 2-5 E and F). Initial values seen from all plots were relatively low and consistent over the first year of the study with small peaks attributed to fertilizer in sample periods 1 and 2, but over time, regardless of fertilizer input, the general trend seen in all plots was increased PO_4^{-3} concentrations in runoff beginning in sample period 10 and continuing throughout the remainder of the study (Fig 2-5 E and F). Concentrations of PO_4^{-3} observed in runoff from lawn types could have serious environmental implications as we observed concentrations above 0.02 mg L^{-1} under every treatment, and these levels could contribute to the eutrophication of surface water bodies.

There were often high concentrations of P seen in runoff samples (Fig 2-5 E and F), but the low levels of runoff that occurred during the given period (Fig 2-5 C and D) which led to low total loading losses from plots (Fig 2-5 G and H). Steinke et al (2007) found similar trends for prairie landscapes versus traditional lawn landscapes in which the higher runoff volume from prairies, especially in the winter months, contributed more significantly to total P loading. Bierman et al (2010) also observed a direct relationship to increased runoff depth with reduced turf quality and growth that led to higher P runoff losses even when no P fertilizer was applied. Similarly in our study, for several nutrients, there were similar or small differences observed in total loading of nutrients from plots that received or did not receive supplemental fertilization, pesticides, and irrigation due to differences in runoff volume. The fertilized turfgrass plots consistently had some of the highest concentrations of P in leachate and runoff samples, however without supplemental irrigation; they had low leaching and runoff volumes. Thus, the effect of nutrient loading was diminished and similar loads were seen from plots that had higher runoff and leaching volumes.

The concentration of P derived by the analysis of TKP in runoff was more variable over time (Fig 2-6 E and F), and there was a significant interaction between treatment and season, treatment and year, and season and year (Table 2-5). Similarly to PO_4^{-3} concentrations, TKP concentrations increased over time for all treatments with the exception of the 100% crabgrass and the turfgrass+irrigation treatments (Fig 2-6 E and F). The interactions were caused by the differing concentrations observed over time among the treatments, which was more drastic for the total contributions of P compared to the orthophosphate contribution in runoff (Fig 2-6 E and F)

The hydrologic concentration of nutrients lost is important to nutrient loading; however the ecological impact can be diminished dependent on the volume of water lost from the system. In our study, we observed a dilution effect for P lost in regards to total loading compared concentrations once we took into effect the total water loss. The most hydrologically important time for the loading of both PO_4^{-3} and TKP was the winter and early spring periods when runoff volumes were highest (Fig 2-5 C and D). The lawn types consisting of turfgrass that received pesticides regardless of irrigation had a significantly higher cumulative load of PO_4^{-3} in runoff compared to all other lawn types (Fig 2-5 G and H; 2-7). However, the total load of P in runoff was highest for the 100% crabgrass and the unfertilized turfgrass plots (Fig 2-7) compared to all other treatments. Neither of these plots received supplemental fertilization and both maintained the lowest overall canopy densities of all other treatments (Fig 2-2) which led to high runoff volumes (Fig 2-5 C and D).

Conclusion

The concentration of P in runoff is one of the most concerning nutrients due to implication its abundance can have in the environment. Phosphorus concentrations were analyzed as both dissolved orthophosphate and total filtered Kjeldahl P to distinguish the between the readily available and more long term sources of P to the environment and try to determine a link between fertilizer applications and natural occurrences. The use of supplemental fertilizers and pesticides to achieve a cosmetically appealing lawn or turfgrass monoculture raises several ecological concerns. The application of fertilization and pesticides can provides a competitive advantage to turfgrass species allowing them to thrive and compete for resources such as light, nutrients, and water, but the application of such supplemental resources has the potential to load the turfgrass ecosystem with nutrients and pesticides that can become mobile if weather presents conditions favorable for runoff losses.

During the course of the study, there were seventy two precipitation events that caused runoff conditions and 68 precipitation events that caused leaching from the study plots. However, discrepancies did occur among the treatments and differences in runoff and leaching volumes we observed. The two major factors measured in the study that appear to most influence runoff volume were the antecedent moisture conditions and vegetative density of the plots. The density of above ground vegetation directly affects the soil moisture content or antecedent conditions by providing differences in plant water use and leaf interception. Those plots that maintained the lowest plant densities consistently had the highest runoff volumes and led to the highest overall loading of P in runoff

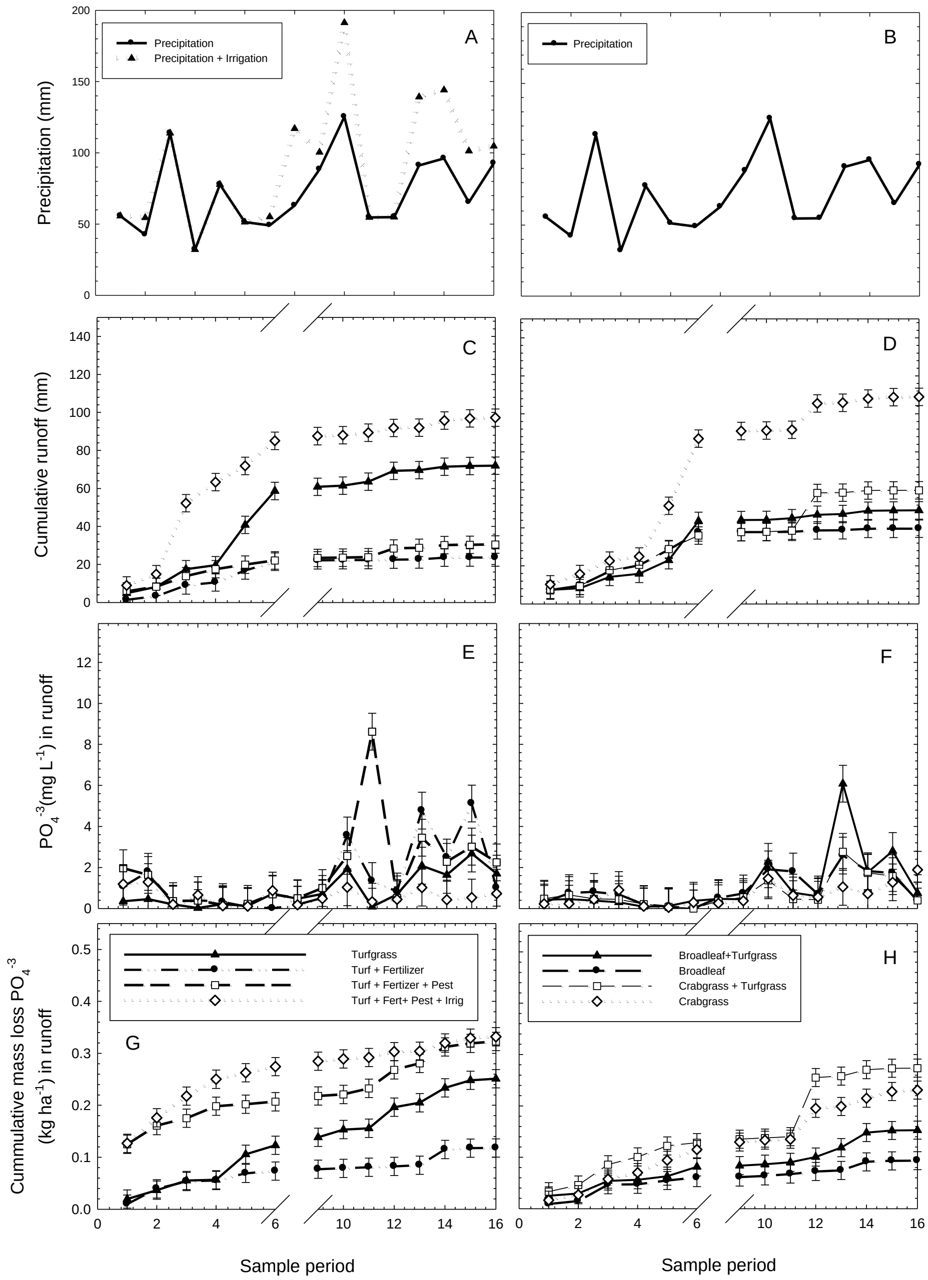

Figure 2-5. Precipitation and irrigation inputs (A)(B), cumulative runoff loss (C)(D), PO_4^{-3} concentrations (E)(F), and cumulative runoff of PO_4^{-3} (G)(H) measured from eight different lawn types through natural precipitation events over 16 sample periods over 27 months. Data break represents a period of renovation in which samples were collected, but no data is available on runoff volumes.

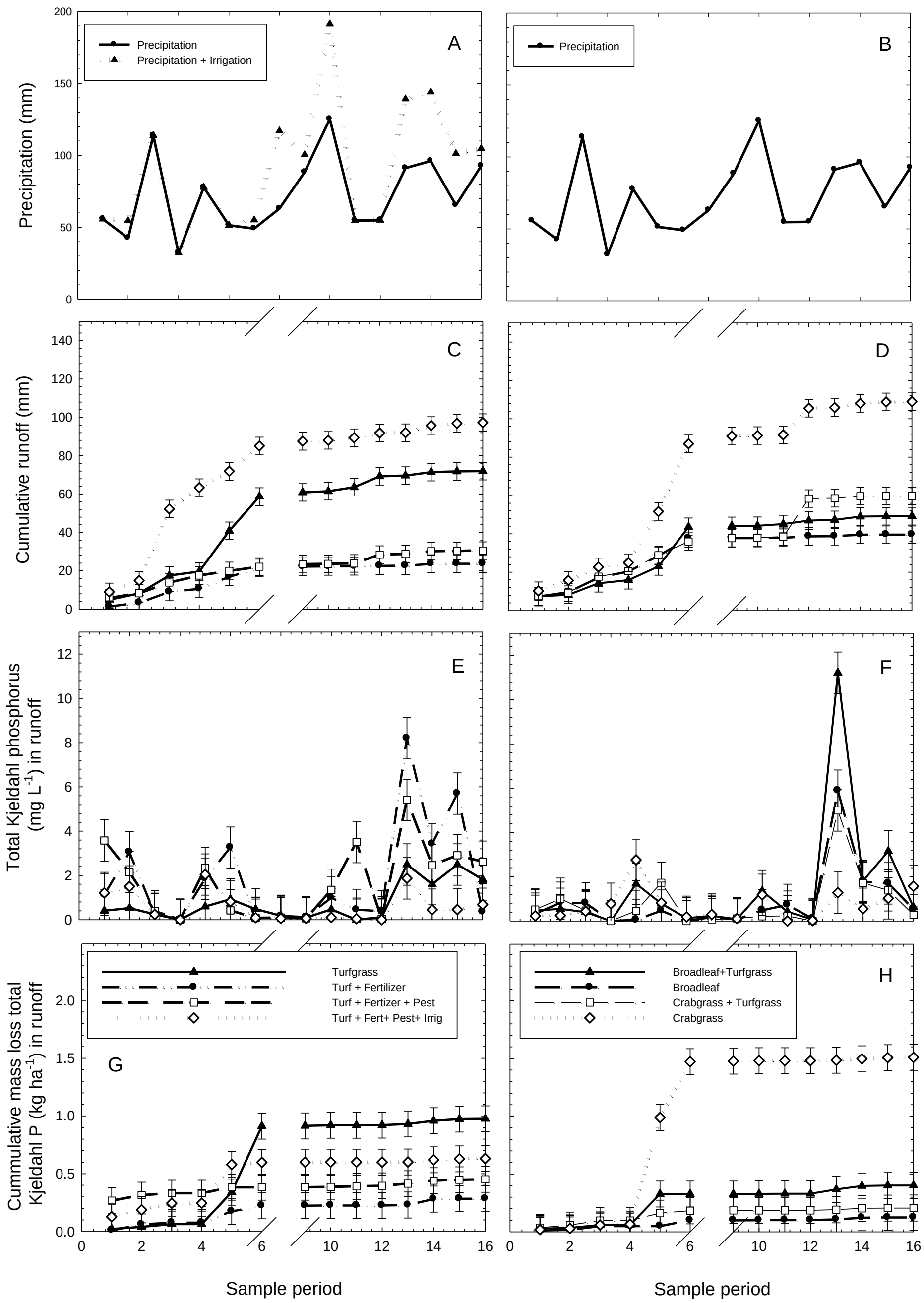

Figure 2-6. Precipitation and irrigation inputs (A)(B), cumulative runoff loss (C)(D), total Kjeldahl P concentrations (E)(F), and cumulative runoff of total Kjeldahl P (G)(H) measured from eight different lawn types through natural precipitation events over 16 sample periods over 27 months. Data break represents a period of renovation in which samples were collected, but no data is available on runoff volumes.

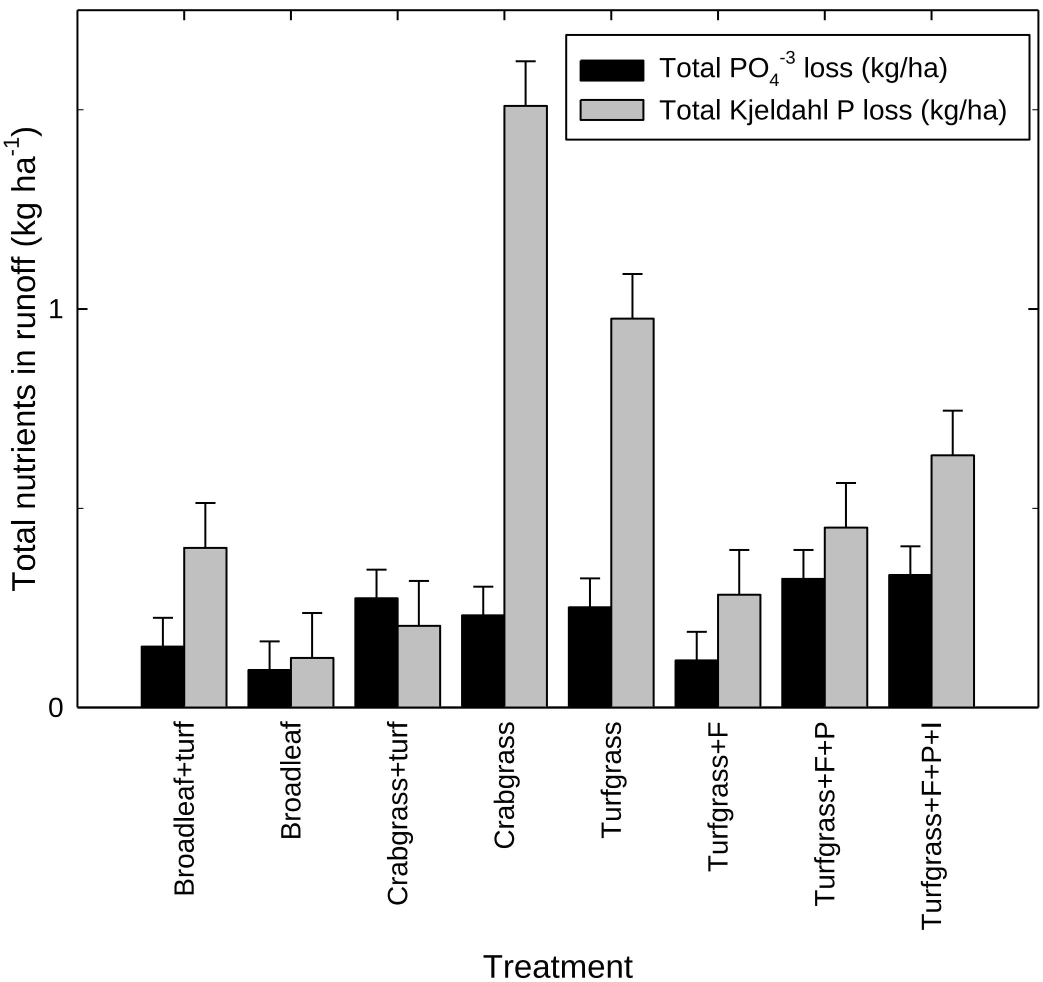

Figure 2-7. Total loading of PO_4^{-3} and total filtered Kjeldahl P in runoff from eight different lawn types over the course of the study.

If soil conditions are kept near saturation, the onset of a runoff event will require less precipitation to saturated the soil and subsequently generate saturation excess storm flow. In areas where adequate natural precipitation occurs for plant needs such as the Northeast of the U.S., supplemental irrigation should be avoided unless periods of drought or below average precipitation occurs, and attempts should be made to maintain dense lawn canopies to reduce nutrient loss whether through supplemental fertilization or through proper cultural practices such as mowing and aeration.

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CHAPTER 3

PESTICIDE FATE IN SODDED KENTCKY BLUEGRASS LAWNS IN RESPONSE TO IRRIGATION

Abstract

The application of fertilizers and pesticides to home lawns can risk contamination of surface and groundwater making them unfit for aquatic organisms and human consumption. However, the application of fertilizers and pesticides for cosmetic purposes not only can improve the aesthetics of a home lawn, but can improve turfgrass shoot and root density, water use, organic matter accumulation, and other factors known to be influential in urban hydrology. Therefore, a field study was set up to determine the impact of applying home landscape pesticides with or without supplemental irrigation to determine the potential for them to leach or runoff to ground or surface water. Free draining lysimeters were place at the top of a slope in a sandy loam soil (coarse-loamy, mixed, active, mesic Lamellic Hapludalf) to monitor the total volume and nutrients and pesticides lost in free draining leachate. Leachate and runoff collected from natural precipitation events were analyzed for ; pendimethalin [N-(1-ethylpropyl)-3,4-dimethyl-2, 6-dinitrobenzenamine] in spring, mecoprop [2-(4-Chloro-2-methylphenoxy)propionic acid] + 2, 4-D (2,4-dichlorophenoxyacetic acid) in late spring and early fall, and bifenthrin [(2-methyl-1,1-biphenyl-3-y1)-methyl-3-(2-chloro-3,3,3-trifluoro-1-propenyl)-2,2-dimethyl cyclopropanecarboxylate]. Pesticide concentrations fluctuated over-time, but no differences in concentrations were observed from irrigated versus non-irrigated turfgrass for any pesticide. 2.4-D and mecoprop were the most mobile forms of pesticides found, and values in runoff, especially during establishment, were above MCL for U.S. water consumption on several occasions. Surface runoff appears to be a more environmentally important pathway for lawn pesticides to move off site as concentrations observed were several

magnitudes higher in runoff compared to leachate for similar time periods. With the exception of one sampling period, pendimethalin concentrations in runoff and leachate were low; however, when applied during a period of minimal turf growth and high precipitation, pendimethalin could contribute significantly to surface water pollution.

Turfgrass often receives scrutiny for its environmental record including water quality near golf courses, sports complexes, and residential sites are questionable. The use of pesticides on turfgrass has a tremendous impact on productivity and quality by reducing the impact weeds, insects, and disease have on color, uniformity and density (Smith and Bridges, 1996). Approximately 27 x 10^6 kg of active pesticide ingredients are applied to turf annually in the U.S. to control damage caused by pests (Petrovic and Larsson-Kovach, 1996).

Cole et al. (1997) summarized several factors that can affect pesticide and nutrient loss including (i) time interval between chemical application and precipitation event; (ii) amount and duration of precipitation event; (iii) antecedent soil moisture; (iv) slope; (v) amount and method of chemical application; (vi) timing of chemical application in regard to plant uptake; (vii) chemical properties; (viii) rate of field degradation/transformation; (ix) soil properties; (x) and vegetation type or density. Studies have shown that dependent on environmental and chemical characteristics, significant portions of pesticides can become mobile in soil solution (Smith and Bridges, 1996; Cole et al, 1997; Starrett et al., 2000). However, Cohen et al. (1999) conducted a study of water quality impacts of golf courses and found no conclusive evidence that turf management on golf courses was causing significant impacts on ground water quality. The pathway of nutrient and pesticide loss can be complicated and related to several environmental and physical factors. Turf systems (verdure, thatch, dense roots) can acts similarly to a living filter and minimize the amounts of pesticides entering surface and groundwater (Baris et al. 2010).

Pesticides can enter surface water through surface transport and groundwater through leaching (Smith and Bridges, 1996; Ma et al., 1999). Pesticide mobility in surface and subsurface flow depends primarily on two factors: water solubility, and soil sorption coefficient (K_{oc}) (Baker and Mickelson, 1994), but not independent of these two factors (Baris et al., 2010). These two values are inversely related so when water solubility increases, the K_{oc} decreases and pesticides will desorb from soil particles more easily by the influx of soil moisture. If a pesticide has high water solubility or a low K_{oc} value, there is an increased risk of ground and surface water contamination. The longer a pesticide persists in the environment; there is an increased chance that it may move off-site (Baris et al., 2010). Thus, a pesticide with high water solubility, low K_{oc} value, and is persistent in the soil would run a higher risk of either surface or ground water contamination through leaching or running off-site.

Most often the highest concentrations of pesticides in runoff and leachate occur during the first rain event following application and concentrations diminish with time. In a simulated rainfall experiment, Smith and Bridges (1996) found that detectable concentrations of mecoprop in a greenhouse lysimeters study only occurred in the leachate for 21d after treatments were applied. For all herbicides [2, 4-D(2,4-dichlorophenoxyacetic acid); mecoprop [2-(4-Chloro-2-methylphenoxy)propionic acid]; dicamba(3,6-dichloro-2-methoxybenzoic acid)], the total analyte transported from the greenhouse lysimeters was less than 1.0% of the herbicide applied. The highest concentrations of herbicides found in leachate did not exceed 6 $\mu g\ L^{-1}$ and the concentrations of dicamba in the leachate from bentgrass did not exceed the minimum detectable level (1 $\mu g\ L^{-1}$). The highest concentrations of 2, 4-D transported from lysimeters containing bentgrass ranged from 3.5 and 5.8 $\mu g\ L^{-1}$ dependent on formulation. The MCL of 2, 4-D of 70 $\mu g\ L^{-1}$ was at least 10 times above the highest concentration determined for the herbicides transported from the lysimeters studies.

Cisar and Snyder (1996) studied the mobility of pesticides to a USGA golf green and found that less than 0.1% of organophosphate pesticides [chlorpyrifos (0,O-diethyl-0-3,5,6-trichloro-2-pyridylphosphorothionte),isazophos [0-[5-chloro-l-(methylethyl)-lff-l,2,4-triazol-3yI] 0,0-diethyl phosphorothioate], isofenphos [l-methylethyl 2-[[ethoxy(l-methylethyl) amino] phosphinothioyl]oxy] benzoate], and ethoprop (0-ethyl 5,5-dipropylphosphorodithioate] applied were recovered in percolate water, regardless of variations in rainfall and total percolation. They concluded that most of the pesticide was retained in the thatch layer and degraded by microbes over time. Chlorpyrifos leaching followed a similar trend when only 0.2% of the total was lost during the first application and only 0.08% was lost during the second application even though the percolation volume increased fivefold. Leaching of isazophos and isofenphos applications resulted in less than 0.1% of total applied for both application dates indicating the relatively low mobility of some pesticides even in a sand based root-zone where infiltration and percolation rates are high. Runoff and leaching losses of pesticides in turfgrass appear to be relatively small. Haith and Rossi (2003) found mean annual pesticide runoff loads did not exceed 3% of the annual applications for any pesticide or site, and applications applied to moss were substantially less than 1% of the application. The risk to surface and subsurface waterways appears to be small when pesticides are applied to turfgrass at appropriate rates and large precipitation events do not directly follow the application dates, but care should be taken when applying pesticides in areas with severe slopes, coarse textured soils, or near waterways to avoid contact with waterways.

In a simulated runoff study on bermudagrass conducted by Smith and Bridges (1996), over 75% of all the herbicides collected in runoff were transported from the plots during the first rainfall event, and only samples collected over the first 192 hours after treatment contained concentrations above the minimum detectable levels. The concentration of 2, 4-D in the runoff water was a factor above the recommended MCL of 70 $\mu g\ L^{-1}$ which suggest the potential for waterway contamination and environmental implications. Runoff and leaching losses of pesticides in turfgrass appear to be relatively small. Haith and Rossi (2003) found mean annual pesticide runoff loads did not exceed 3% of the annual applications for any pesticide or site, and applications applied to moss were substantially less than 1% of the application.

Objective

The objective of this study was to examine the potential losses of commonly applied lawn pesticides used to control insects and broadleaf and grassy weeds in response to irrigation on Kentucky bluegrass (*Poa pratensis L*.). The goal is to determine differences in hydrology of plots based on supplemental irrigation and how it affects the hydrological mobility of commonly used pesticides in the landscape.

Materials and Methods

The following study was conducted at the Turf and Landscape Research Center at Cornell University in Ithaca, NY. A southwest facing hill-slope having a slope range of 13-15% and measuring 39 m by 6.1 m was stripped of all sod to a depth of 3.8 cm. Topsoil was removed with a bulldozer and stockpiled for later use. The subsoil was removed to a depth of 45 cm and the underground subsoil was thoroughly compacted with the bulldozer by repeatedly driving over the site. The remaining subsoil and topsoil was replaced and compacted in 15 cm lifts with the bulldozer to ensure consistent compaction levels and to provide conditions similar to a new home construction site.

The hill slope was subdivided into eight 1.2 m wide x 6.1 m long plots. Lysimeters were constructed similar to Miltner et al. (1996) made of polyvinyl chloride (PVC) pipe with a 61.0 cm diameter and 91.4 cm length. Each length of PVC was driven into the ground vertically in the top 1.2 m section of the hill slope, using a loader until the top of the lysimeters was flush with the surface of the subsoil. All attempts were made to remove the lysimeters with the soil core intact, and once removed, a four inch gravel layer was placed at the bottom, and a flat sheet of 12.7 mm thick PVC with a diameter of 61.0 cm was attached to the base with PVC cement to ensure a tight seal. If a soil core was not removed intact, the soil was removed in 15 cm lifts and placed in the lysimeters in the same sequence it was removed, and then tamped to ensure the same quantity of soil fit into the lysimeters to maintain the bulk density of the soil profile. Approximately one half of the lysimeters or 2 reps were removed with the core intact and the other half were removed by hand. Once all bases were affixed to the lysimeters, they were lowered back into the excavated site they were removed from.

A trench 1.0 m deep was placed on the downhill slope of the lysimeters to give access to the base and a 12.7 mm diameter exit drain was placed at the bottom of each lysimeters. Polyethylene pipe with a diameter of 12.7 mm was connected to each exit drain with a barbed fitting and tubing from eight lysimeters was run downhill to a common collection area for leachate sampling in 4000 mL plastic containers. After the installation of the lysimeters, trenches were backfilled and compacted to prevent settling. The entire area was covered in approximately 7-8 cm of topsoil and hand raked in preparation for establishments.

The study consisted of a randomized complete block design with the variation in hill-slope used as the blocking factor to provide four blocks with two treatments per block. Plastic landscape edging was placed between each plot to a depth of 50 mm to confine flow of surface water within plots. Treatments consisted of Kentucky bluegrass (*Poa pratensis* L.) (KB) lawn that received cosmetic applications of fertilizers (F) and pesticides (P) using consumer lawn products (the Scotts-Miracle

Gro Co., Marysville, OH) that provide 4 fertilizer applications and 4 pesticide applications; pendimethalin [N-(1-ethylpropyl)-3,4-dimethyl-2, 6-dinitrobenzenamine] in spring, mecoprop [2-(4-Chloro-2-methylphenoxy)propionic acid] + 2, 4-D (2,4-dichlorophenoxyacetic acid) in late spring and early fall, and bifenthrin [(2-methyl-1,1-biphenyl-3-y1)-methyl-3-(2-chloro-3,3,3-trifluoro-1-propenyl)-2,2-dimethyl cyclopropanecarboxylate] in summer, and supplemental irrigation (I). Irrigation was applied when soil moisture content decreased below 20% relative water content which was determined to be the level of 50% depletion of plant available water (Brady and Weil, 1999). Plots were irrigated with 1.25 cm of water at each irrigation event with a below ground irrigation system with the ability to supply irrigation at a rate of 75 mm hr^{-1}. The volume of irrigation water applied did not provide adequate irrigation to induce runoff losses of irrigation from the plots. The second treatment was the same as the first described above, but without supplemental irrigation. All plots were mowed weekly at a height of 5 cm with a rotary mower similar to those used in a home lawn and all clippings were removed. Plots received supplemental fertilization and pesticides according to a seasonal schedule described in Table 3-1. Plots were sodded on 3 July 2007 with sod containing four different cultivars (Odyssey, Liberator, Rugby II, and Total Eclipse). All plots were irrigated to facilitate root establishment and prevent wilt during establishment.

During establishment, a 155 mm wide by 200 mm deep trench was placed at the bottom of all plots running perpendicular to the hill slope. A 127 mm wide, 1.20 m long channel drain gutter was cut to the length of each plot and placed with a 0.5% slope at the base of each plot with a 12.7 mm diameter drain placed at one end connected to polyethylene pipe that led the water to a collection site for the surface runoff water. All containment gutters were covered with 3.2 mm thick metal sheeting to stop rain and outside contaminants from entering the collection site. The drains for the channel gutters ran downhill where piping was grouped for every two plots collection area where soil was excavated and a 95 L plastic storage container was placed in the soil. The polyethylene pipe entered through the uphill side of the container and was extended over 100 mL tipping buckets. Tipping buckets were leveled and placed on 12.7 mm thick plywood with holes drilled large enough for 1000 mL sampling bottles to be placed below the buckets. Tipping buckets were connected to event data loggers (Onset Computer Corp., Pocasset, MA, model HOBO® pendent event logger) and every tip was calibrated prior to use by pouring 1000 mL of water into each collection gutter periodically and measuring the number of tips for the given volume. The number of tips and frequency of tips were measured for each storm event and logged to equate total runoff losses from each plot. Runoff storm events were monitored and classified as a precipitation event when runoff occurred to any plot within the study.

Storm events that produced leachate and runoff were separated by precipitation event; however, often multiple storms occurred within several hours to days of each other and leachate volumes could not be separated from events. In such a case, total leachate volumes were recorded for the entire precipitation event and the event was labeled as a single precipitation event with the precipitation amounts recorded for the entire period. Subsamples of leachate and runoff were taken within 6 h of storm events, and frozen immediately until further analysis. Subsamples were compiled into sample periods based on the frequency of storms in a time period to obtain 3-6 storms per period and to reduce the number of samples to analyze. Subsamples were bulked proportionally to the volume of leachate for the given storm event to create bulked subsamples with nutrient concentrations proportionate to the actual storm events.

Samples were analyzed for the presence of all pesticides applied at the Cornell University Analytical Chemistry Laboratory, New York State Agricultural Experiment

Station Geneva, NY 14456. Pendimethalin [N-(1-ethylpropyl)-3,4-dimethyl-2, 6-dinitrobenzenamine], and bifenthrin, and were analyzed using gas chromatography - electron capture detection (GC/ECD) after the addition of 1 mL of 1.5N HCl to 25g of water sample. The samples were mixed, and 20 mL of Methylene chloride ($MeCl_2$) and 5g of NaCl were added and shaken for 30 minutes. The samples were then transferred to a separatory funnel and rinsed with an additional 5 mL of $MeCl_2$. After 30 seconds of shaking, layers were allowed to separate and the lower ($MeCl_2$) layer was transferred to a graduated centrifuge tube where samples were evaporated to dryness under nitrogen in a warm (40°C) water bath. After evaporation, samples were brought up with 2 mL of toluene.

Samples were analyzed for 2, 4-D (2,4-dichlorophenoxyacetic acid) using GC/ECD by extracting 25 mL of the sample water in a 125 mL separatory funnel (with recovery spike added if appropriate) and shaken for one minute with three successive 5 mL portions of $MeCl_2$ added. Methylene chloride was evaporated under nitrogen in a warm water bath and residues were brought up in 2 mL of ethyl acetate. 2-4,-D residues were analyzed by methods outlined by Cessna et al. (1985). Mecoprop [2-(4-Chloro-2-methylphenoxy) propionic acid] (MCPP) was analyzed using high pressure liquid chromatography (HPLC) after evaporating samples under nitrogen in a water bath (40°C). Samples were brought up with 2 mL of a 60:40 acetonitrile: water solution then filtered through 0.45µ filter for HPLC analysis described by Petrovic and Larsson–Kovach (1996).

Table 3-1. Application dates and product descriptions of the fertilizer and pesticide combinations applied to the Kentucky bluegrass turfgrass plot

Application date	Nitrogen applied	Phosphorus applied	Potassium applied	Active ingredient	
	kg ha^{-1}				g kg^{-1} applied
3 September 2007	36.46	1.72	8.65	Bifenthrin	0.86
24 September 2007	30.07	1.52	11.52	2,4-D; Mecoprop	12.1; 6.1
23 April 2008	41.17	1.81	4.56	Pendimethalin	12.2
26 May 2008	39.15	1.84	3.48	2,4-D; Mecoprop	12.1; 6.1
28 July 2008	36.46	1.72	8.65	Bifenthrin	0.86
3 October 2008	30.07	1.52	11.52	2,4-D; Mecoprop	12.1; 6.1
29 April 2009	41.17	1.81	4.56	Pendimethalin	12.2
26 May 2009	39.15	1.84	3.48	2,4-D; Mecoprop	12.1; 6.1
22 July 2009	36.46	1.72	8.65	Bifenthrin	0.86
16 September 2009	30.07	1.52	11.52	2,4-D; Mecoprop	12.1; 6.1
Total applied	**360.23**	**17.04**	**76.59**		

Volumetric moisture content of plots were taken periodically using time domain reflectometry (TDR) through a water content probe (Spectrum Technologies Inc., Plainfield, IL, model: FieldScout TDR 300) with 12 cm rods to monitor irrigation frequency and antecedent conditions. Measurements were taken at the top, middle, and bottom of each plot to quantify dynamics that may affect water movement above or below the soil surface.

The study was initiated on 10 August, 2007 when the first supplemental irrigation event was applied to KB plots receiving irrigation plus pesticides and fertilizer applications and storm events were monitored for 27 months through 1 November, 2009. During the second winter of the study, frost tubes (McCool and Molnau, 1984) were place in the center of each plot to determine the relationships between cover type and soil freezing, and also the relationship between soil freezing depth and runoff and leaching frequencies. Data was monitored primarily during the initial soil freezing period or when climatic conditions were favorable for snowmelt or rainfall that may induce leaching.

On 2 September 2008, 7.3 cm diameter undisturbed soil cores were taken 6 cm deep from each plot. A soil core was taken from the top, middle, and bottom portion of each plot and stored in a cooler at 4°C until a complete physical analysis could be completed. Samples were analyzed for distribution of pore space, penetrometer resistance, and bulk density as described in (Moebius et al, 2007).

Soil cores measuring 2.2 cm diameter x 8 cm deep were extracted on 26 September 2007, 21 October 2008, 26 June 2009 and 12 August 2009 from several random locations within each plot and homogenized to create a soil sample representative of each plot. Samples were used to determine soil particle distribution through the hydrometer method (Sheldrick and Wang, 1993), Morgan extractable phosphorus (Morgan, 1941), soil pH in 0.01 M $CaCl_2$ (Hendershot et al, 1993):

organic matter from loss on ignition (Karam, 1993).

Pre-established soil infiltration potentials were measured within each block on 23 May 2007 using a miniature rain infiltrometer (Ogden et al. 1997) and secondary infiltrations measurements were taken on 5 June 2008 and 23 September 2009 within each plot to monitor changes in infiltration through the establishment period.

Statistical analysis

All leaching and runoff pesticide data was subjected to analysis of variance using a mixed model in JMP 8.0 (SAS institute, 2009) with repeated measures. Treatment and sample period were treated as fixed effects, while the blocking factor was treated as a random variable in the model. When significant differences were observed, means were separated using Tukey's HSD with $\alpha = 0.05$ probability level.

Soil sample and plot characteristics were subjected to ANOVA with repeated measures with the exception of undisturbed soil core data that was taken only once. If following ANOVA, there was no significant effect of time on a measurement, data for the plot was averaged over time and means were separated using Tukey's HSD test with an $\alpha = 0.05$ probability level when significant effects were observed through ANOVA.

Results and Discussion

There were important hydrological differences observed among irrigated and non-irrigated treatments. The application of irrigation to Kentucky bluegrass led to significantly higher portions of precipitation occurring as both leachate (Fig. 3-1 A) and runoff (Fig. 3-1 B). The relationship is quite distinct and seasonal based on the sampling periods from which runoff and leachate occurred. The largest differences in runoff and leachate discrepancies occurred during the first eight sample periods which represented the first year of the study (Table 3-2). Furthermore, leachate volumes were significantly higher under irrigated conditions throughout the course of the study

(Fig. 3-1 A).   As expected the primary difference observed between the lawns with and without irrigation was the higher soil VWC seen under irrigated KB (Fig. 3-2) along with an increasing soil moisture gradient from the top of the plots to the bottom. The higher VWC observed under irrigated conditions led to an increased potential for soil saturation that resulted in higher leachate volumes and also higher rates of saturation excess runoff to occur. The leachate volumes were consistently higher under irrigated conditions for the entire course of the study (Fig 3-1 A). The runoff volumes observed were significantly higher under irrigated conditions for the first year of the study, after which there were no differences among the two treatments for the remainder of the study (Fig 3-1 B).

Pesticides in leachate

Four pesticides were applied to the KB lawns as part of the preventative pest management program. Water solubility and K_{oc} values of these pesticides vary and are described in Table 3-3.   There were no significant differences observed in the concentrations of any pesticide under irrigated versus non-irrigated KB throughout the study in either runoff or leachate and therefore all data was averaged among the two treatments. Concentrations of pesticides found in leachate varied over time, but higher concentrations were seen directly following applications of the fertilizers containing pesticides (Fig 3-3) and diminished with time. Overall, the transport of these pesticides in leachate to depths that could potentially contaminate subsurface waterways was minimal.

Table 3-2. Leaching and runoff events and corresponding date ranges assigned to sample periods

Sample period	Date range	Leachate events	Date range	Runoff events
1	8 Sept 2007-15 Sept 2007	1-3	8 Sept 2007-15 Sept 2007	1-4
2	16 Sept 2007-27 Oct 2007	4-9	16 Sept 2007-27 Oct 2007	5-13
3	28 Oct 2007- 3 Dec 2007	10-16	28 Oct 2007- 3 Dec 2007	14-18
4	4 Dec 2007-11 Jan 2008	17-23	4 Dec 2007-11 Jan 2008	19-21
5	12 Jan 2008- 29 Feb 2008	24-30	12 Jan 2008- 29 Feb 2008	22-28
6	1 Mar 2008- 4 Apr 2008	31-37	1 Mar 2008- 4 April 2008	29-33
7	5 Apr 2008- 17 May 2008	38-39	5 Apr 2008- 17 May 2008	34-35
8	18 May 2008- 15 July 2008	40-41	18 May 2008- 15 July 2008	36-40
9	16 July 2008- 10 Aug 2008	42-44	16 July 2008- 10 Aug 2008	41-45
10	11 Aug 2008- 15 Nov 2008	45-47	11 Aug 2008- 15 Nov2008	46-50
11	16 Nov 2008- 5 Jan 2009	48-51	16 Nov 2008- 5 Jan 2009	51-53
12	6 Jan 2009- 30 Mar 2009	52-55	6 Jan 2009- 30 Mar 2009	54-57
13	31 Mar 2009- 6 May 2009	56-57	31 Mar 2009- 18 June 2009	58-62
14	7 May 2009- 30 June 2009	58-61	19 June 2009-26 July 2009	63-66
15	1 July 2009- 29 Aug 2009	62-64	27 July 2009- 29 Aug 2009	67-72
16	30 Aug 2009- 15 Nov 2009	65-68	30 Aug 2009- 15 Nov 2009	73-75

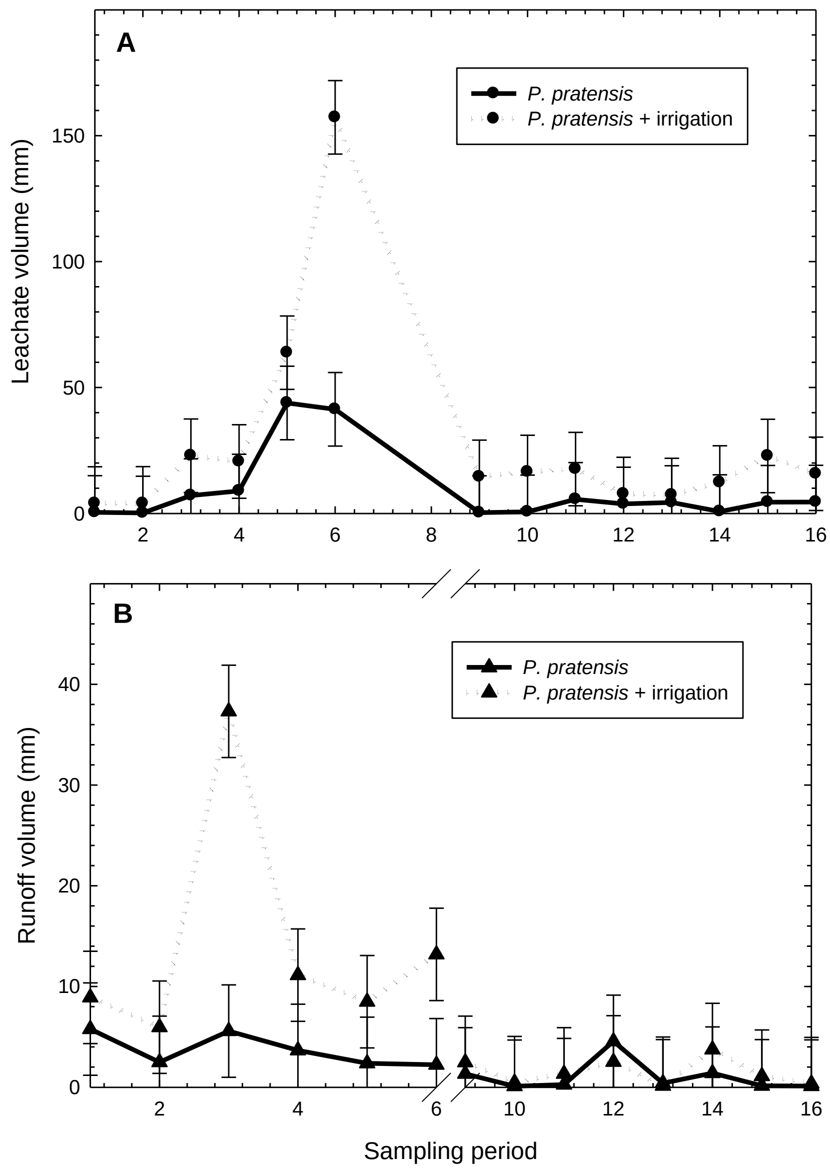

Figure 3-1. The observed differences in leachate volume (A) and runoff volume (B) from natural precipitation among irrigated and non-irrigated Kentucky bluegrass turfgrass over a 27 month period. Data break in runoff volume is due to plot renovations. During this period samples were taken and analyzed for concentrations; however, no runoff volumes were recorded.

In our study, mecoprop was the most mobile in soil solution and was found in the greatest concentrations in the leachate, contradictory to what was expected based on K_{oc} values and water solubility (Table 3-3). Concentration values did not exceed 20.0 µg L^{-1} throughout the entire study, but did often exceed the U.S EPA water standard MCL of 10 µg L^{-1} during several sampling periods (Fig 3-3). The largest potential loss of mecoprop in leachate was in sampling periods following the late spring application for broadleaf weed control. This is also the period of time when there were large leachate volume losses from the plots (Fig 3-1). Pendimethalin, bifenthrin, and 2,4-D values in leachate were low and concentrations measured were below the water standards U.S. EPA water standards (Table 3-3) and did not show potential for loading significantly to subsurface waterways at any time during the study (Fig 3-3).

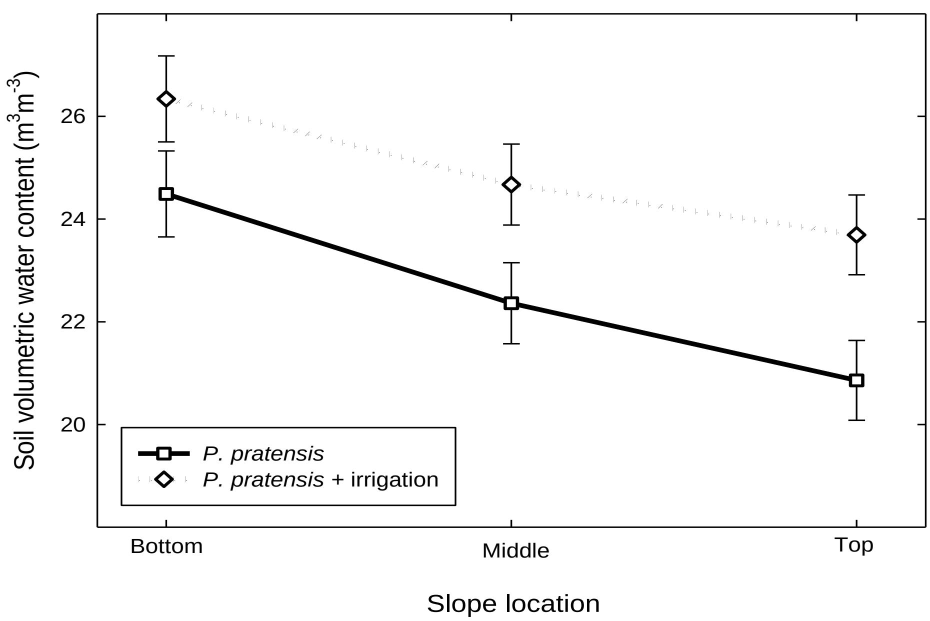

Figure 3-2. The average soil VWC measured over several dates throughout the growing season under irrigated and non-irrigated Kentucky bluegrass turfgrass.

Table 3-3. Environmental characteristics of four pesticides used in care and maintenance of turfgrass lawn type treatment

Pesticide	Type	Water† solubility	Adsorption coefficient	Aerobic soil half-life	U.S. water standard MCL
		mg L^{-1}	**K_{oc}**	**days**	**µg L^{-1}**
Bifenthrin	Insecticide	0.10	237,000	95	n/a
Pendimethalin	Pre-emergent herbicide	0.30	5000	40	20.0
2,4-D	Post-emergent herbicide	27,600	46.0	34.0	70.0
Mecoprop	Post-emergent herbicide	869	119	20	10.0

† http://www.pesticidedata.org

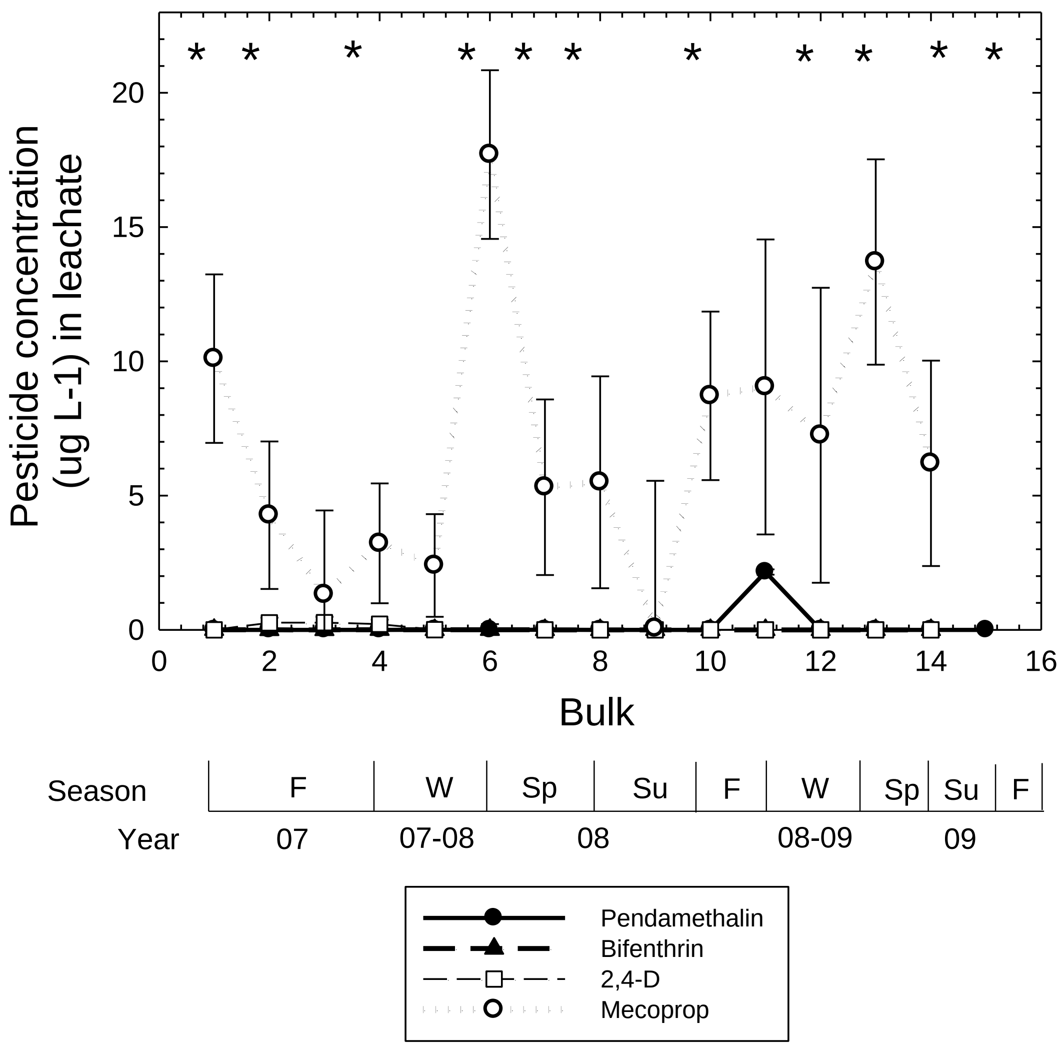

Figure 3-3. Mean concentrations of four common landscape pesticides found in leachate over time. Due to insignificant differences related to irrigation, values were averaged among treatments that received pesticides with or without irrigation. Bars represent the standard error of the mean.

* pesticide application timing

Pesticides in runoff

There was no difference observed in concentrations of pesticides seen in runoff under irrigated or non-irrigated turfgrass suggesting that an irrigation application designed to meet plant needs, will not increase the risk of pesticide runoff. Therefore, data on runoff concentrations was averaged for the two irrigation treatments (Fig 3-4). There was a significant effect of sample period on the pesticide concentrations found in runoff, and the establishment period was seen to be an especially critical period in which pesticides could be lost from the environment. There were large peaks in concentrations of both mecoprop and 2, 4-D herbicides in the first fall of the study directly following the first application of the products (Fig 3-4). During initial soil infiltration measurements for the plots during establishments, infiltration capacities were <1cm hr^{-1}, but over time increased dramatically to >10 cm hr^{-1} for both lawn treatments through the course of the study. This factor influenced the runoff volumes and concentrations of pesticides found in the runoff samples. Beyond the initial fall and for the remainder of the study, runoff concentrations of 2, 4-D and mecoprop were lower with the exception of substantial concentrations of both herbicides observed the first late spring application (Fig 3-4). The values at both of these sampling events were several magnitudes greater than the U.S. water standard MCL for the herbicides (Table 3-3) and could provide potential harm to aquatic organisms or humans if lost near surface waters.

Runoff samples analyzed for bifenthrin and pendimethalin were below maximum detection limits (0.001 mg L^{-1}) for the majority of the study, with one large peak of increased pendimethalin concentrations observed within the spring application and during sample period 6 (Fig 3-3).

The mobility of pesticides in runoff is directly related to the water solubility of the products and hydrologic mobility of 2,4-D and mecoprop was apparent throughout the study as runoff concentrations were substantially higher for these two pesticides.

Conclusion

The hydrological mobility of pesticides applied to the landscape is concerning for aquatic, vegetative and human health. Differing chemistries of pesticides along with cultural practices applied to the landscape are factors that can impact the transport of herbicides to surface and groundwater from application sites. The process of applying supplemental irrigation to a home lawn has been indicated as a potential mechanism to reduce surface loss of lawn applied pesticides by washing material off of leaves and incorporating them into the soil. This process could reduce the risk of losses through runoff inducing precipitation events by incorporating products into the soil; allowing for faster plant root uptake or soil immobilization.

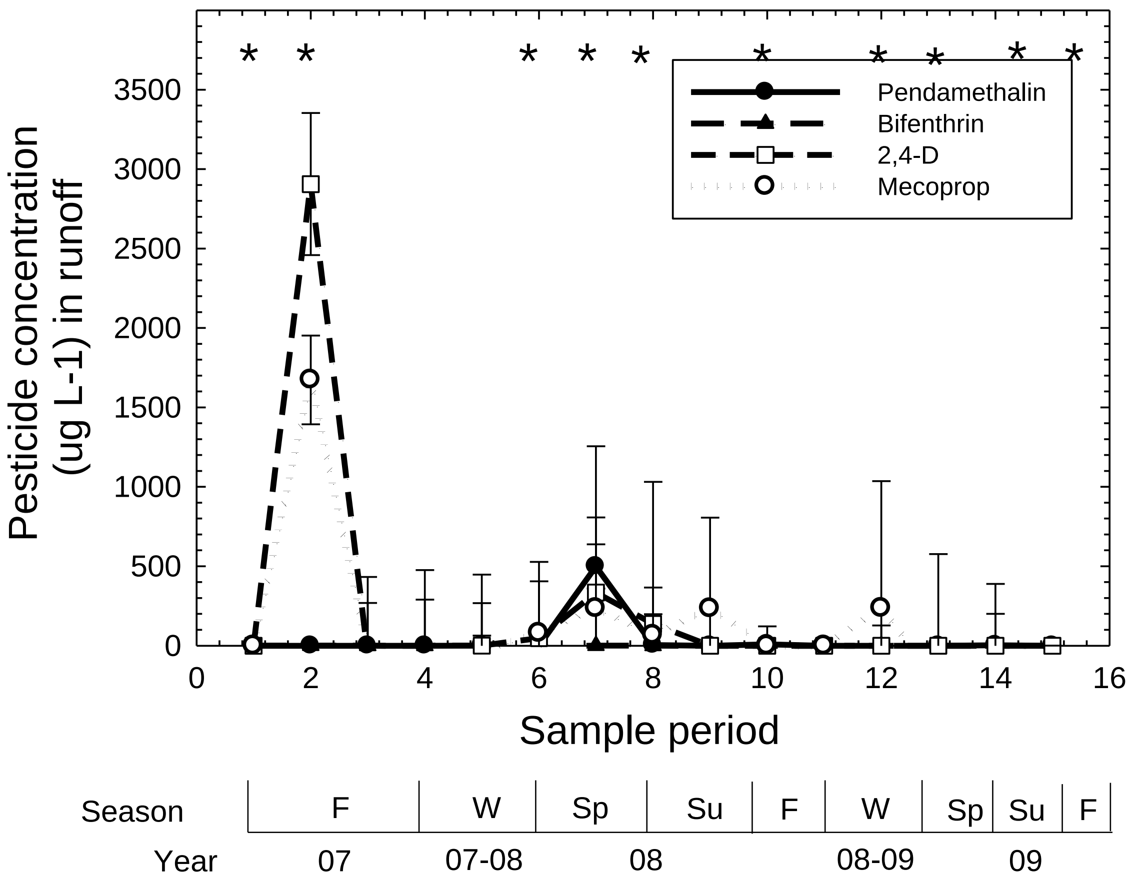

Figure 3-4. Concentrations of four common lawn pesticides found in runoff from Kentucky bluegrass lawn types over a 27 month period.

In our study, there were trends towards lower concentrations of pesticides in runoff and leachate under irrigated conditions; however, the differences were not statistically significant. The climate of Ithaca, NY is one that receives substantial natural precipitation throughout the growing season, and throughout the course of the study, plots were only irrigated 6-8 times per season. Supplemental irrigation at no time improved the visual density or color of the plots beyond that of the non-irrigated plots and may not have been necessary in the region to improve growth and density beyond what was achieved through natural rainfall. However, supplemental irrigation did maintain a higher average soil VWC than non-irrigated conditions which led to higher leachate and runoff volume losses. This phenomenon could result in higher losses of pesticides; regardless of lower concentrations, because the potential for higher total loading could exist.

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CHAPTER 4

TURF QUALITY AND NUTRIENT LEACHING FROM *AGROSTIS STOLONIFERA* L. GROWN IN ORGANIC AMENDED SAND WITH HIGH-NUTRIENT CONTAINING BIOCHAR AND SPHAGNUM PEAT

Abstract

The establishment period for turfgrass is an important period in which applied nutrients fate can become hydrologically mobile without root uptake and plant interception. Sand-based root zones have minimal micropore space and are more prone to nutrient leaching; therefore are often amended with organic matter to improve nutrient and water retention. A greenhouse study was designed to monitor the effect a nutrient rich biochar amendment could influence creeping bentgrass (*Agrostis stolonifera* L.)(CB) germination and quality while monitoring the nutrient leaching potential compared to non-amended and sphagnum peat moss amended sand root zones. Free draining lysimeters were filled with sand or sand amended with peat moss or biochar at a 90:10 v v^{-1} ratio and was either provide supplemental fertilization at a rate of 4.9 kg N ha^{-1} or no supplemental fertilization. Seeded CB was allowed to establish and monitored for density and quality while periodic leaching events were forced through applying 9.3 mm of irrigation beyond saturated soil conditions to flush available nutrients from the rootzone. Both quality and above ground density were significantly improved through the use of biochar as an amendment under fertilized and unfertilized treatments. Nitrate leaching was significantly higher under fertilized sand and biochar amended sand plots during the initial leaching periods of both trials, but levels decreased to those of peat moss containers over-time. Phosphate concentrations in leachate for biochar amended sand under fertilized and unfertilized conditions were significantly higher than all other treatments, and linearly increased with time over both trials. Phosphorus contributions in biochar amended sand from

the organic material were significantly higher, and direct linear relationship was established with the Morgan extractable P levels in the soil to the phosphate levels in runoff. Biochar has the potential to improve soil fertility and CEC of the soil beyond peat moss and provide conditions that are beneficial for germination and quality of CB; however, the use of a nutrient rich feedstock could result in groundwater contamination of N and P beyond values seen from peat moss.

Introduction

The establishment of turfgrass is a critical period both environmentally and economically. It can be a time of high nutrient demand for young, rapidly developing plants, and can also pose a risk of nutrient, pesticide and sediment loss through leaching and runoff. During the establishment period, large amounts of surface soil are exposed to the environment, potentially leading to losses of soil and sediments through wind and surface erosion (Deletic, 2001; Deletic, 2005; Easton and Petrovic, 2004). Eroded soil particles can transport plant essential nutrients from establishing turfgrass sites to waterways (Gross et al., 1991, Krenitsky et al., 1998; Sharpley et al. 1992). These sediments can supply nitrogen (N) and phosphorus (P)(Carpenter et al., 1998; Corell, 1998; Sharpley et al., 1992, Sharpley et al., 2000), to aquatic organisms where they often are limited. Once nutrients become available to algae and microbes, they can produce algal blooms that deplete the surface waters of oxygen (Carpenter et al., 1998; Sharpley et al., 1992)

Nitrogen leaching from turfgrass has been shown to be a function of soil texture, fertilizer management and irrigation practices (Petrovic, 1990). The use of amendments in sand-based root zone construction and the study of nutrient losses have been minimally studied (Paré et al., 2008; Pare et al., 2006; Huang and Petrovic, 1994). Paré et al (2008) found that one-third to one-half of the NO_3^- leached from a sand-peat moss profile was derived from N accumulated in the soil, presumably from mineralized organic N, which should be a factor in determining fertilizer requirements. Amending sand root zones with inorganic soil conditioners such as natural zeolite have shown to reduce NO_3^- and NH_4^+ leaching through increased sorption and uptake compared to sand alone (Huang and Petrovic, 1994). Nitrate concentration in leachate averaged between 13.5 to 26.5 mg N L^{-1} depending on root architecture from a sand-based pot trial seeded with creeping bentgrass (Bowman et al., 1998) which exceeded the EPA drinking water standard of 10 mg L^{-1} (Carpenter et al., 1998). Bowman et al. (2002) found that that NO_3^--N and NH_4^+-N readily leached from six warm season species of turfgrass studied in sand based root zones under greenhouse conditions, but nutrient leaching declined substantially as the root system developed.

Phosphorus leaching is generally a minor concern in natural settings because most soils have high P sorption properties (Corell, 1998). Under four natural grassland species, Turner and Haygarth (2000) found total P export in leachate to be <0.5 kg ha^{-1} yr^{-1} which represented <1% of the annual fertilizer applied. Barton et al. (2006) found the total P losses from Bermudagrass leachate were low (< 4 kg P ha^{-1} after 22 months) representing less than 1.5% of the applied phosphorus. King et al (2006) found that subsurface drainage of P from two golf course sites in tile drains were greater than measurements from tile drains in agriculture and could pose a significant environmental threat while NO_3^- levels were approximately one-tenth the concentration from agriculture. However, sands generally are characterized as having low P sorption properties (Soldat and Petrovic, 2008), which combined with fertilizer and irrigation inputs may lead to significant P loss in leachate (Mathers et al., 2007).

Research has shown that one of the most influential factors in reducing off-site sediment and nutrient loading to waterways is to provide a dense plant canopy (Easton and Petrovic, 2004; Cole et al., 1997); therefore providing ways to either establish or maintain plant density quickly would be advantageous to water quality.

The typical golf course green root structure, as suggested by the United States Golf Association (USGA)(USGA Green Section Staff, 1993), is a sand-based root zone modified with either organic or non-organic amendments to enhance establishment and to improve the nutrient and water holding capacity above sands natural characteristics. Traditional amendments consist of, but are not limited to; sphagnum peat moss, composts, and zeolite. Most of the amendments used are from non-renewable or slowly renewable resources, and providing a sustainable replacement option for those amendments could be environmentally advantageous.

Biochar is a soil amendment that has the potential to replace typical sand-based root zone amendments in turfgrass systems while offering environmentally sustaining properties. Biochar is a carbon (C) rich product obtained through pyrolysis when biomass, such as wood, manure, leaves or turfgrass clippings are heated in a closed container with little or no available oxygen. The production processes for biochar, along with its intended use as a soil application, distinguish it from charcoal, which is one of the most ancient industrial technologies developed by mankind (Harris, 1999). Biochar may affect a multitude of soil properties and processes such as cation exchange capacity (Sombroek et al., 1993; Liang et al., 2006), nutrient dynamics (Lehmann et al., 2003), and microbial habitation (Pietikäinen et al., 2000). Chan et al. (2007) found a significant biochar and fertilizer interaction that led to higher yields of radish (*Raphanus sativus* var. Long Scarlet) induced through higher N use efficiency when soils where amended with biochar.

Biochar is beneficial as a soil amendment mainly due to its high organic C content and porous morphology. Black carbon sources such as biochar are more recalcitrant in nature compared with other carbon pools making them an important component of the global carbon cycle (Zimmerman, 2010; Lehmann et al., 2008) Often, traditional organic amendments applied to sand-based turfgrass systems offer short term benefits through increased cation exchange capacity and water holding potential, but provide little or no mineralizable nutrients due to high C:N ratios.

The objective of this study was to examine the use of biochar as an alternative to traditional organic amendments in sand based root zone construction for turfgrass establishment by measuring the impact to creeping bentgrass (*Agrostis stolonifera* L.) establishment and growth. Secondary to this objective is to identify potential environmental implications in using a grass or green-type feed stock that is high in nutrients as an organic amendment to improve crop growth and reduce leaching of plant essential nutrients.

Materials and Methods

Three soil mixtures consisting of either straight quartzite sand, sand mixed with biochar or sand mixed with sphagnum peat moss at a 90:10 (v v^{-1}) ratio were placed in free-draining lysimeters 0.2 m in diameter and 0.38 m deep with a 50 mm gravel layer at the bottom. The sand's physical properties were within the standards of the USGA recommendations for golf course green construction (USGA Green Section Staff, 1993) and had a bulk density of 1.23 g cm^{-3}. The addition of biochar and peat moss as amendments to the sand reduced the bulk density to 1.15 and 1.18 g cm^{-3} respectively. In addition to changes in bulk density through organic amendment additions, total N and P inputs from sphagnum peat moss and biochar were 6.40 and 18.88 g N $container^{-1}$ and 0.21 and 4.61 g P $container^{-1}$ respectively. The biochar was created through slow pyrolysis at a temperature of 500º C using grass clippings from a golf course putting green consisting of a mixture of creeping bentgrass (*Agrostis stolonifera* L.) and annual bluegrass (*Poa annua* L.)

At the base of each lysimeters, a 12.7 mm plastic tube was placed to facilitate the free draining of leachate to 500 mL collection bottles. Lysimeters were established with Penn A-1 creeping bentgrass (*Agrostis stolonifera* L.) on 5 December

2008 for trial 1, which was conducted through 23 March 2009. The second repeated trial was established on 5 August 2009 and ran through 30 November 2009. Pots for both trials were seeded at a rate of 98 kg ha^{-1}. Treatments consisted of either a weekly fertilization to the containers, or no supplemental fertilization, creating a 3 x 2 factorial completely randomized experimental design with four replications (n=24). Following the first trial, all media from the pots was homogenized within the respective mixes. This combined all soil from the fertilized and unfertilized treatments to randomly assigned lysimeters for the second trial in order to reduce the carryover of any effects associated with the first trial.

The experiment was carried out in a temperature controlled glass house maintained at average night time temperatures of 15.5˚C and average daytime temperatures of 24˚C. The photoperiod was set to 16 h d^{-1} and was supplemented with high pressure sodium lamps. The container water holding capacity for each lysimeter was established by measuring the lysimeters mass after the free draining of gravitational water following irrigation (Huang and Petrovic, 1994). Pots were misted with 5 mm of water twice d^{-1} for three weeks to maintain moist surface conditions for germination and establishment. Following initial germination, to obtain relative water use differences among treatments, lysimeters were irrigated three days a week by obtaining the lysimeters mass and adding irrigation until established container capacity was reached. Pots were mowed at approximately 7 mm 3 d wk^{-1} prior to irrigating to maintain mowing conditions similar to a golf course putting green grow in.

Photos and quality measurements were taken with a digital camera (8 Megapixel, Model-Sure shot, Canon Inc.) periodically to monitor establishment rates and overall quality. Photos were analyzed digitally for percent green cover (Karcher and Richardson, 2003) and quality ratings were taken during the first trial based on a

1-9 scale with 6 being the minimum acceptable quality (Skogley and Sawyer, 1992). Lysimeters that were fertilized, received fertilizer on a 7 d interval at a rate of 4.9 kg N ha^{-1} containing 50% urea and 50% ammonium nitrogen sources, and phosphorus (P_2O_5) and potassium (K_2O) at matching rates of 1.27 kg ha^{-1}.

Leaching events were induced by bringing each lysimeters to established container capacity, at which point 9.3 mm of supplemental irrigation was added to each plot to force gravimetric water loss as leachate. This provided the opportunity to monitor the potential loss of nitrate (NO_3^-), ammonium (NH_4^+), and orthophosphate (PO_4^{-3}) during establishment. Leachate was collected in 60-mL subsamples and frozen immediately until analyzed.

Because leachate was forced after each individual container was brought to container capacity, the value of nutrients analyzed in leachate is presented as nutrient concentrations. This method allowed for consistent leachate volumes to be forced among containers because leaching directly following the saturating the soil to container capacity; therefore, relative differences that typically would be reported as mass loss or mass loading of nutrients based on volume differences would not be relevant since all containers contained similar leachate volumes.

Leachate nitrate was measured colorimetrically after reducing NO_3^- to NO_2^- (EPA method 114-A) through a copperized cadmium coil. NO_2^- was measured colorimetrically at 520 nm using a spectrophotometer. The first group of samples were analyzed for NH_4^+-N using (EPA method 351.2) by reacting ammonia with alkaline salicylate and hypochlorite to form an analog of indophenol blue that was measured colorimetrically at 650-660 nm. Ammonium levels for all treatments were below test detection limits for the first two dates of leachate sampled, therefore only samples from those events in trial 1 were analyzed for the presence of ammonium in leachate and no results will be further discussed on ammonium leaching and N will refer solely to NO_3^- nitrogen. Phosphorus was measured colorimetrically as orthophosphate (EPA method 365.1) through the reaction of acidic molybdate in the presence of antimony to form an antimony phospho-molybdate complex which is reduced by ascorbic acid to a blue complex measured colormetrically at 660 nm.

Plant above ground biomass and below ground biomass were sampled after trial 1 to determine a relationship between digital image density and dry weight as well as to determine soil amendment affects on root and shoot dynamics of the turfgrass. Soil cores measuring 22 mm diameter x 80 mm deep were taken from the containers at the end of trial 2 and analyzed for Morgan extractable phosphorus (Morgan, 1941) to correlate P in leachate with P content of the soils.

Saturated hydraulic conductivity (K_{sat}) was measured *in Situ* for the amended soil types by maintaining a constant 10 mm head of water on top of the soil surface while monitoring the efflux of water from the exit drain until steady state conditions were obtained (Reynolds, 1993).

Cation exchange capacity (CEC) was established for the sand, peat moss amended and biochar amended soil mixtures through the extraction of base cations through the ammonium acetate method (Hendershot et al., 1993). Plant tissue was analyzed through dry ash of tissue and analysis through Inductively Coupled Plasma Atomic Emission Spectrometry (ICP/AES) for macro and micronutrients while nitrogen was determined through dry combustion.

Statistical analysis

All data was subjected to analysis of variance using a generalized linear model in JMP 8.0 (SAS Institute, Cary, N.C.). When significant treatment affects were observed, means were compared using Tukey's honestly significant difference for main effects of soil amendment and fertilization as well as their interaction. Unless otherwise noted, significant differences observed are reported at the $p < 0.05$ level.

Due to significant differences in the two experimental trials over-time, responses in trial 1 and 2 were analyzed separately.

Results

Physical and chemical characteristics

Biochar amended profiles provided different characteristics, both physically and chemically (Table 4-1). Biochar provided a C:N ratio of 10.8 which was more than a fourfold lower ratio than the peat moss amendment. The adsorptive capacity of biochar was much higher than peat moss, and after CEC determination using ammonium acetate, it was determined that the peat moss, biochar, and sand had CEC's of 185.88, 1645.86, and 261.71 $mmol_c\ kg^{-1}$ respectively (Table 4-1). The differences in CEC and the C:N ratios of the sand and amendments can provide important differences in nutrient retention and nutrient cycling. The total contributions of N and P were significantly different among amendments, and based on the mass of the amendment for the volume integrated, biochar contributed 4.61 g P and 18.88 g N $container^{-1}$ which provide 22 and 3 times the rate of N and P respectively than the sphagnum peat moss provided.

There were significant differences observed in the saturated hydraulic conductivity (K_{sat}) of the soil profiles (Figure 4-1). Non-fertilized peat moss containers had higher conductivity than the fertilized peat moss due to surface crusting and algal growth that occurred on the fertilized container. There was no difference observed among the remaining treatments, and all treatments maintained a K_{sat} value well above the standard of the USGA green profile of 15 $cm\ hr^{-1}$ desired to promote rapid downward moisture movement; however, our lysimeters were not maintained under compacted conditions similar to the field.

With the exception of a time effect, there were no significant differences in the use of water among the container types in trial 1 (Table 4-2). In trial 2, there was a significant three way interaction between media type, fertilizer, and date ($p = 0.026$) in which the unfertilized biochar and peat moss containers used significantly less water per day than the unfertilized sand containers (Table 4-2). However, because plant densities and biomass were so low in trial 2 among the unfertilized pots, these differences are relatively insignificant to the overall scope of the study. The interaction revealed no differences between the fertilized containers of the three soil types in the second trial.

Visual quality and density

The highest overall visual quality ratings were achieved by the fertilized biochar pots (Fig. 4-2). Averaged over the entire study period for trial 1, they maintained a rating of 5.3 which was significantly higher than fertilized peat moss and sand containers which maintained quality ratings of 4.3 and 4.2 respectively. Variation in trials over time occurred in the treatments due to an outbreak of disease (*Pythium* spp.) during the three week seedling stage that reduced the above ground biomass. In both trials, biochar pots that received fertilization produced significantly denser turfgrass ($p = 0.05$) compared to peat moss and sand pots (Fig 4-3). In trial 1, the fertilized biochar pots reached peak density approximately 45 days into the trial, while the sand pots took approximately 90 days to reach a similar density. The fertilized peat moss soil pots never reached an established density > 60% under the growing conditions. Biochar pots that received no supplemental fertilization also produced significantly higher ($p = 0.05$) plant densities and overall turf qualities compared to non-fertilized peat and sand pots in trial 1. The fertilized biochar containers in trial 2 once again produced significantly denser turf, but due to disease outbreak, only a 56% density was achieved for the study. These containers were the only containers to significantly recover from the *Pythium* outbreak while the fertilized sand and peat moss had minimal densities (Fig. 4-3b).

Root and shoot relationship

The final shoot dry and root dry mass data was collected at the end of trial 1. There were significant differences seen for shoot dry mass, root dry mass and percent root mass among fertilizer and soil treatments (Table 4-3). There was a benefit of fertilization seen for the amended soils that were fertilized, but the same benefit to shoot mass was not seen for the fertilized sand pots. The shoot mass for the fertilized containers containing biochar and peat moss were significantly larger than all unfertilized containers, and the fertilized biochar shoot mass obtained was significantly larger than the shoot masses obtained in the fertilized sand containers (Table 4-3).

The shoot and root biomass produced among the treatments are important, but the percent of resources allocated to root biomass can be an important indicator of carbohydrate allocation. Among the treatments imposed on the grass species, only the fertilized sand and unfertilized biochar containers had significantly higher percent root mass allocation than the unfertilized peat moss containers which had the lowest percent root mass allocation (Table 4-3). Not only was the shoot biomass for unfertilized peat moss containers the lowest, but the allocation of biomass among the *A. stolonifera* species to root biomass rather than shoot biomass was also the lowest of all the treatments (Table 4-3). Evaluation of plant tissue nutrient concentrations revealed some important trends and explanations to differences in root and shoot growth along with density and visual quality (Tables 4-4 and 4-5). In regards to tissue nutrient contents, the fertilized biochar plots always had the same or significantly higher portions of both macro and micro plant essential nutrients than the sand and peat moss amended soils. There was also significantly higher tissue P found in the tissues grown in the unfertilized biochar containers compared to the fertilized peat moss containers. This finding suggests an important nutrient availability component with the addition of nutrient rich biochar amendments beyond that received from sphagnum peat moss.

Nutrient Leaching

Nitrate

There were significant differences in NO_3^--N concentrations observed in leachate over time (Fig. 4-4). With the exception of one leaching event in trial 1 where sand and biochar leached significantly more NO_3^-, concentrations were similar for all fertilized pots and greatly diminished over time as turf density increased (Fig. 4-4). Both the sand and biochar containers that were fertilized had leachate concentrations of NO_3^- up to 60 mg L^{-1} 31 days after seeding (DAS), while the fertilized peat containers had a peak N leachate concentration of 29 mg L^{-1} on the first leaching event that occurred 27 days before seeding. Leaching of the unfertilized treatments in trial 1 followed a similar trend, but to a smaller magnitude. Leachate forced from unfertilized biochar pots 31 DAS had concentrations of 24 mg L^{-1} while concentrations from the unfertilized sand and peat moss pots never exceeded concentrations of 10 mg L^{-1} at any date..

Table 4-1. Chemical characteristics of material used in root zone mixtures for establishing *A. stolonifera*.

	Total N	**Total C**	**Total P**	**C:N**	**C:P**	**N:P**	**pH**	**CEC**
	mg g^{-1}			**ratio**				**$mmol_c\ kg^{-1}$**
Biochar	49.0	535.0	11.96	10.8	44.7	4.1	9.6	1645.86
Sphagnum peat moss	9.0	414.0	0.30	47	1380	30	4	185.88
Sand	n/a	n/a	n/a	n/a	n/a	n/a	n/a	261.71

Table 4-2. ANOVA table of the effect of media type, fertilizer and date on the daily water use of *A. stolonifera* in two different trials repeated in time.

		Trial 1		Trial 2	
Source	***df***	**F ratio**	**Prob > F**	**F ratio**	**Prob > F**
Media	2	0.198	ns†	5.63	**
Fertilizer	1	0.001	ns	4.10	*
Date	34	1.191	ns	32.46	***
Media x Fertilizer	2	0.835	ns	12.63	***
Media x Date	68	0.991	ns	0.82	ns
Fertilizer x Date	34	0.971	ns	0.58	ns
Media x Fertilizer x Date	68	0.975	ns	1.59	*

†Significant at *α=0.05, **α=0.01, and ***α=0.001; ns=not significant

Table 4-3. Shoot dry mass, root dry mass and percent root mass allocations of *A. stolonifera* L. grown in three different media types with and without supplemental fertilization

		Shoot dry mass	Root dry mass	Fraction root mass
	Media	**g pot^{-1}**		**g g^{-1}**
Fertilized	Biochar	35.39 a†	17.71 ab	0.384 ab
	Peat	27.73 ab	21.74 a	0.427 ab
	Sand	14.34 bc	15.54 abc	0.516 a
Unfertilized	Biochar	10.59 c	8.91 bcd	0.482 a
	Peat	2.77 c	0.61 d	0.256 b
	Sand	4.31 c	5.72 cd	0.432 ab

† Values within a column with different letters are significantly different at the α = 0.05 probability level

Table 4-4. Plant tissue macronutrient contents obtained at harvest from *A. stolonifera* plants grown in quartzite sand, or sand amended with biochar or sphagnum peat moss under fertilized and unfertilized conditions.

		N	**P**	**K**	**Ca**	**Mg**	**S**
	Media			**g kg^{-1}**			
Fertilized	Biochar	7.08 ab†	2.10 a	10.25 a	10.98 ns	2.84 ns	2.48 ab
	Peatmoss	4.08 b	1.15 b	5.18 b	9.28	2.41	2.14 b
	Sand	8.18 a	1.83 ab	8.70 ab	10.90	3.09	3.17 a
Unfertilized	Biochar	5.65 ab	2.22 a	7.68 ab	11.60	3.12	1.91 b
	Peatmoss	5.25 ab	1.41 ab	5.90 b	13.03	2.89	1.51 b
	Sand	5.78 ab	1.57 ab	6.90 ab	11.18	2.94	1.69 b

† Values within a column with different letters are significantly different at the $\alpha = 0.05$ probability level
ns = values within a column are not significantly different

Table 4-5. Plant tissue micronutrient contents obtained at harvest from *A. stolonifera* plants grown in quartzite sand, or sand amended with biochar or sphagnum peat moss under fertilized and unfertilized conditions.

		B	Fe	Mn	Cu	Zn	Al
	Media			**mg kg^{-1}**			
Fertilized	Biochar	11.3 a†	5985.35 a	189.68 a	46.63 ab	99.93 a	5609.08 a
	Peatmoss	8.98 ab	5591.00 ab	109.23 b	55.60 a	73.35 b	4493.20 ab
	Sand	9.05 ab	3879.25 b	120.25 ab	42.93 ab	84.98 ab	2923.08 c
Unfertilized	Biochar	10.5 ab	5261.33 ab	168.48 ab	47.25 ab	89.63 ab	4106.85 bc
	Peatmoss	8.13 b	4029.40 b	139.68 ab	38.80 b	72.58 b	3103.10 bc
	Sand	9.8a b	5312.35 ab	163.18 ab	47.73 ab	76.98 ab	4359.23 abc

† Values within a column with different letters are significantly different at the $\alpha = 0.05$ probability level

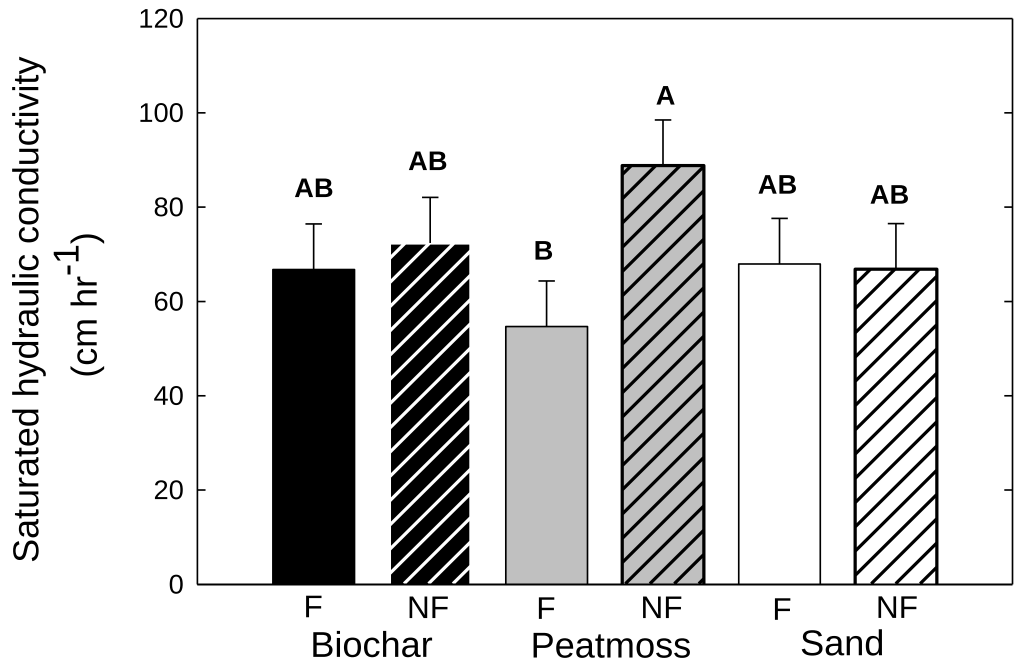

Figure 4-1. *In Situ* hydraulic conductivity of sand-based, sand amended with peat moss, and sand amended with biochar media types under fertilized (F) or non fertilized (NF) conditions. Bars containing different letters represent significant differences at α=0.05 level.

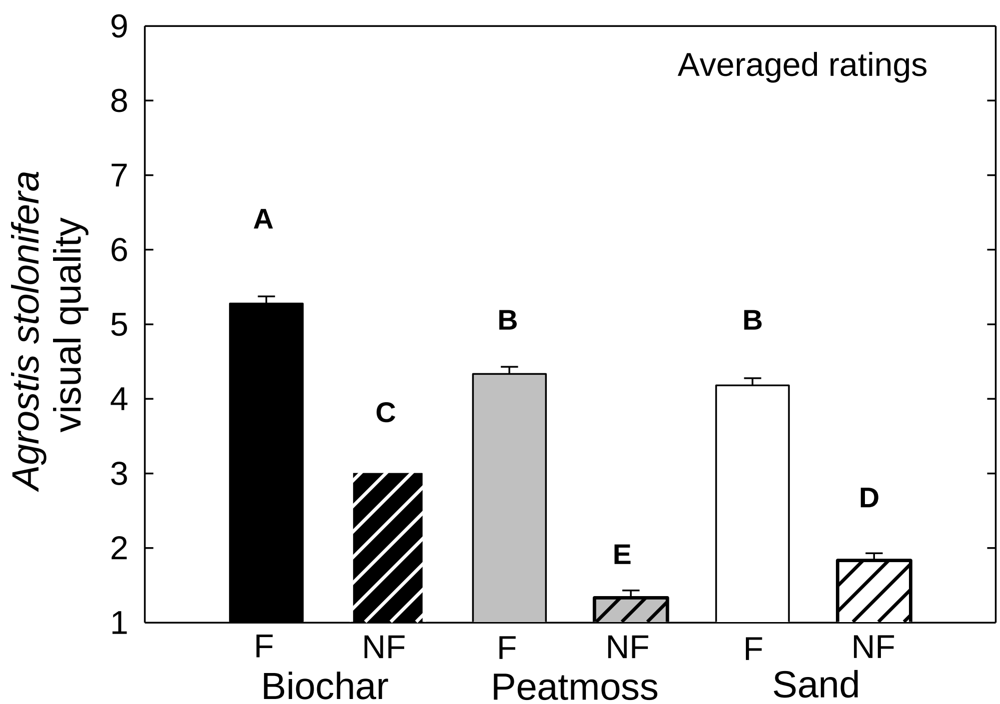

Figure 4-2. Overall visual quality ratings averaged over time (1-9 scale: 9=best) of creeping bentgrass (*Agrostis stolonifera* L.) grown in three different soil amendments during trial 1, under fertilized (F) and unfertilized (NF) conditions. Bars with different letters represent significant treatment differences ($\alpha = 0.05$) for the combined data consisting of all measurement dates

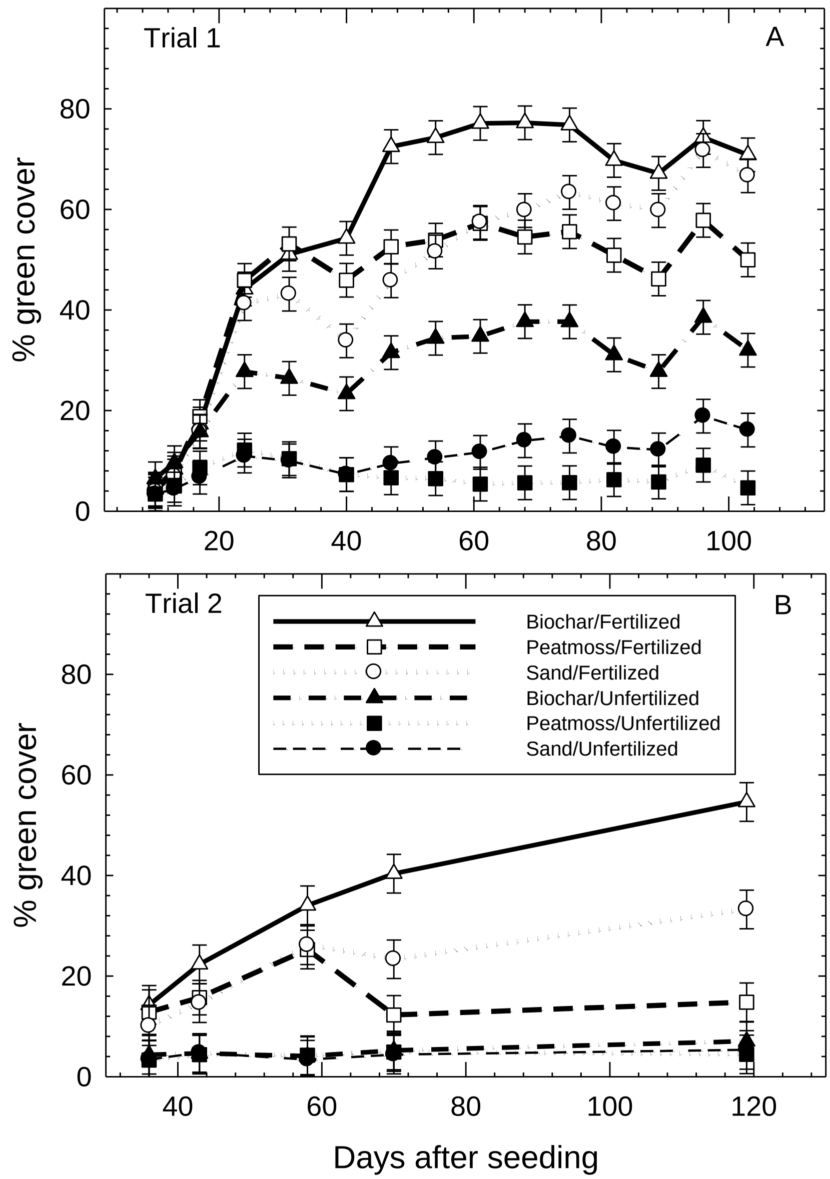

Figure 4-3. Percent cover derived from digital image analysis during trial 1(a) and trial 2 (b) taken at periodic intervals for three soil types and two fertilizer treatments. Images were less frequent during trial 2 due to an outbreak of Pythium disease (14 DAS) which greatly decreased the rate of growth for all treatment combinations.

During trial 2, NO_3^- leachate concentrations were significantly higher for both sand and biochar than for peat moss (Fig. 4-4) similarly to trial 1, but the magnitude of concentration in leachate was significantly higher. The sand and biochar containers that were fertilized both had peak N concentrations of 99.8 and 106.2 mg L^{-1} respectively occurring 40 DAS while the peak NO_3^- concentrations seen in the peat moss pots was 38 mg L^{-1} seen 57 DAS (Fig. 4-4). The concentrations observed in the unfertilized sand and biochar were also significantly higher in the second trial compared to the first, and both were significantly higher than the peat moss container overall (Fig. 4-4). Although large peaks in concentrations were observed, over time the general trend of NO_3^- concentrations in leachate decreased substantially in both trials as plant densities increased.

Phosphorus Leaching

Phosphorous concentrations were significantly higher for fertilized and unfertilized biochar pots across all time points compared to all other treatments for trial 1 and trial 2 (Fig. 4-5). Concentrations among fertilized and non-fertilized biochar were not significantly different from one another, attributing the concentrations of P to the addition of biochar and not fertilizer inputs. The general trend for both trials was for leachate concentrations of P to increase over-time in the biochar containers, while the sand and peat moss maintained relatively stable. Low P concentrations were seen in trial 1 for sand and peat moss pots (<0.4 mg L^{-1}) however, in trial 2, the P concentration in the sand and peat moss containers began to increase over time (Fig. 4-5). Under fertilized conditions, sand and peat moss showed concentrations of 1.028 and 0.644 mg P L^{-1}, respectively, and 1.291 and 0.551 mg P L^{-1}, respectively, under unfertilized conditions (Fig. 4-5).

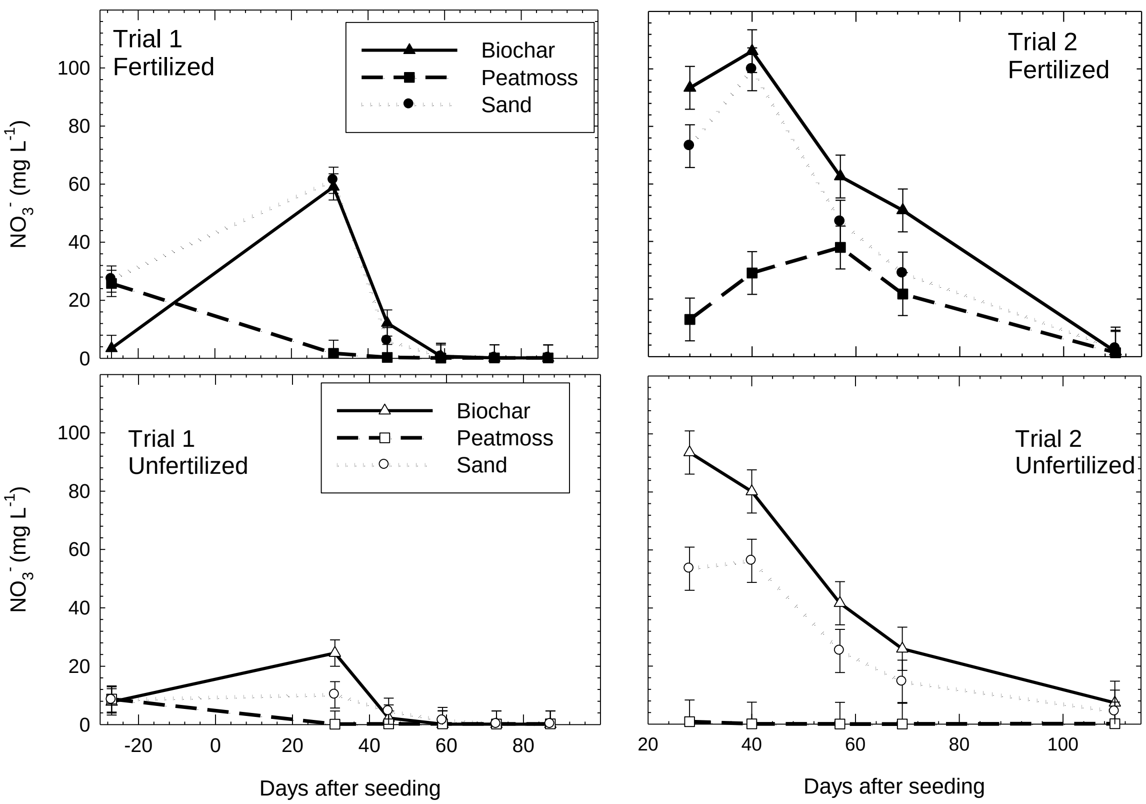

Figure 4-4. Nitrate N concentrations in leachate during trial 1 and trial 2 after leachate was forced with 9.3 mm of water beyond container capacity at periodic intervals. The first data points for trial 1 correspond to a baseline leaching event prior to any fertilizer applications. Unfertilized pots in trial 2 received supplemental fertilizer for two weeks to aid in disease recovery 14 DAS.

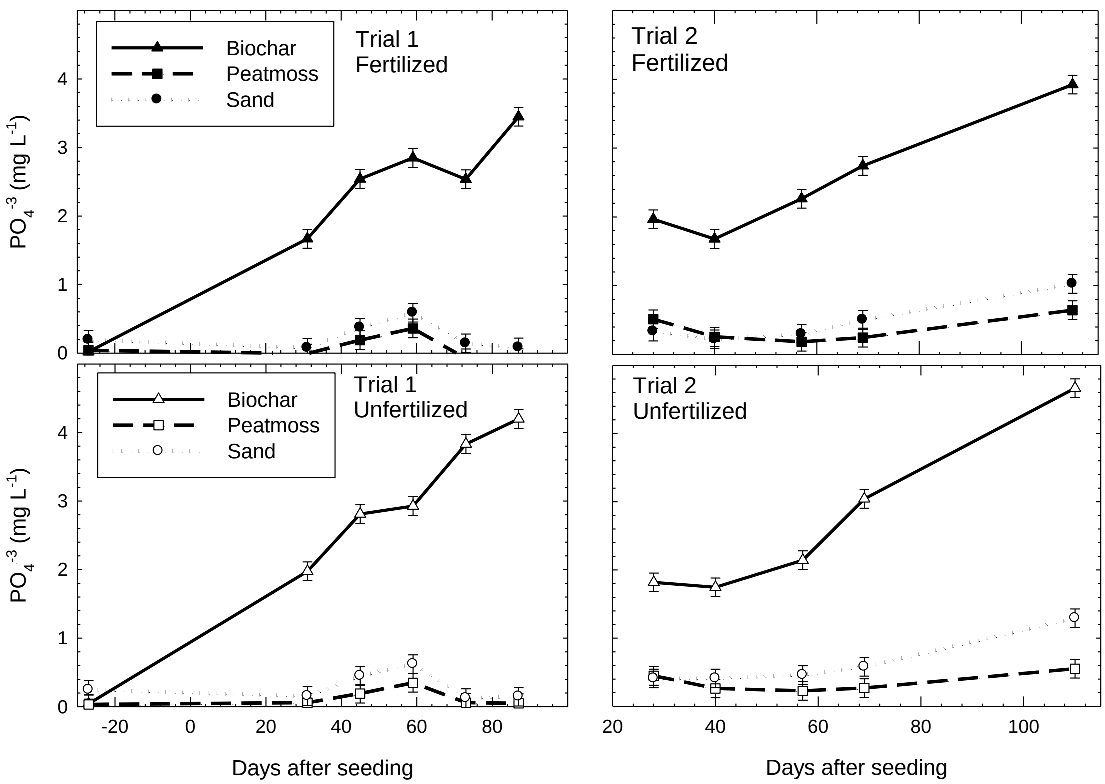

Figure 4-5. Orthophosphate P concentration in leachate during trial 1 and trial 2 after leachate was forced with 9.3 mm of water beyond container capacity at sampling times. The first data points in trial 1 correspond to a baseline leaching event prior to any fertilizer applications. Unfertilized pots received supplemental fertilizer during trial 2 for two weeks to aid in disease recovery 14 DAS.

Phosphorus concentrations in leachate taken prior to soil sampling were strongly correlated (r^2=0.879) to the Morgan extractable phosphorus levels in the soil based on a linear regression equation (Fig. 4-6). Phosphorus levels in the biochar pots were significantly higher by two to three times than the extractable P levels in the sand and peat moss containers. The addition of fertilization to the containers significantly increased the extractable phosphorus levels among all soil types; however that difference was only expressed in higher P leachate concentration in the biochar amended pots. With the addition of the biochar amended soil, total P contributions were twenty two times higher than sphagnum peat moss amended containers. Therefore, differences in release and concentrations of phosphorus in leachate can effectively be explained by the potentially mineralizable phosphorus pool in the biochar amended sand over that of the other two soil types.

Discussion

Many products appear to be good alternatives to traditional organic amendments but many negatively affect soil physical characteristics. The use of biochar derived from turfgrass clippings did not reduce hydraulic conductivity or appear to negatively affect surface conditions. During this study, initial nitrate N concentrations did occur in the biochar amended sand and straight sand pots, but levels decreased to those similar to peat moss amended sand over time. Once plants became established and developed root zones capable of intercepting and taking up more nutrients around 40-60 DAS (Fig. 4-4 and 4-5), nitrate concentrations in leachate diminished. It appears that peat moss amended sand may provide a better media for retarding the leaching of nitrate from the root zone than biochar. The difference in N leaching may also be due to the significantly higher C:N ratio of peat moss compared to biochar. The higher ratio of carbon to nitrogen in peat moss provides little opportunity for the mineralization of organic N in the root zone (Bragazza et al., 2007;

Hue and Sobieszczyk, 1999), thus minimizing the pool of N available to leach from the profile. Although potentially beneficial from a water quality standpoint, the lack of available N in the root zone caused by the higher C:N ratio, may contribute to the low foliar N concentrations and slower establishment of turfgrasses seen in peat moss amended containers (Hue and Sobieszczyk, 1999). The higher CEC levels observed in biochar versus peatmoss and sand (Table 4-1), biochar amended sand may provide surface conditions better able to retain cations such as NH_4^+ making them more available longer term.

Phosphorus leaching could be a concern if plants are established during periods of high precipitation and/or irrigation. There was a clear distinction and trend of biochar amended containers to leach significantly higher levels of phosphorus, regardless of fertilizer applications. For most lakes, streams, reservoirs, and estuaries, concentrations of 100 μg total P L^{-1} are unacceptably high and concentrations of 20 μg L^{-1} are often a problem (Corell, 1998), and the values observed in this study were well above those levels. However, while lysimeter studies are important for comparing treatment effects, results are not necessarily a proxy of absolute amounts that will be lost from an environmental setting under natural conditions.

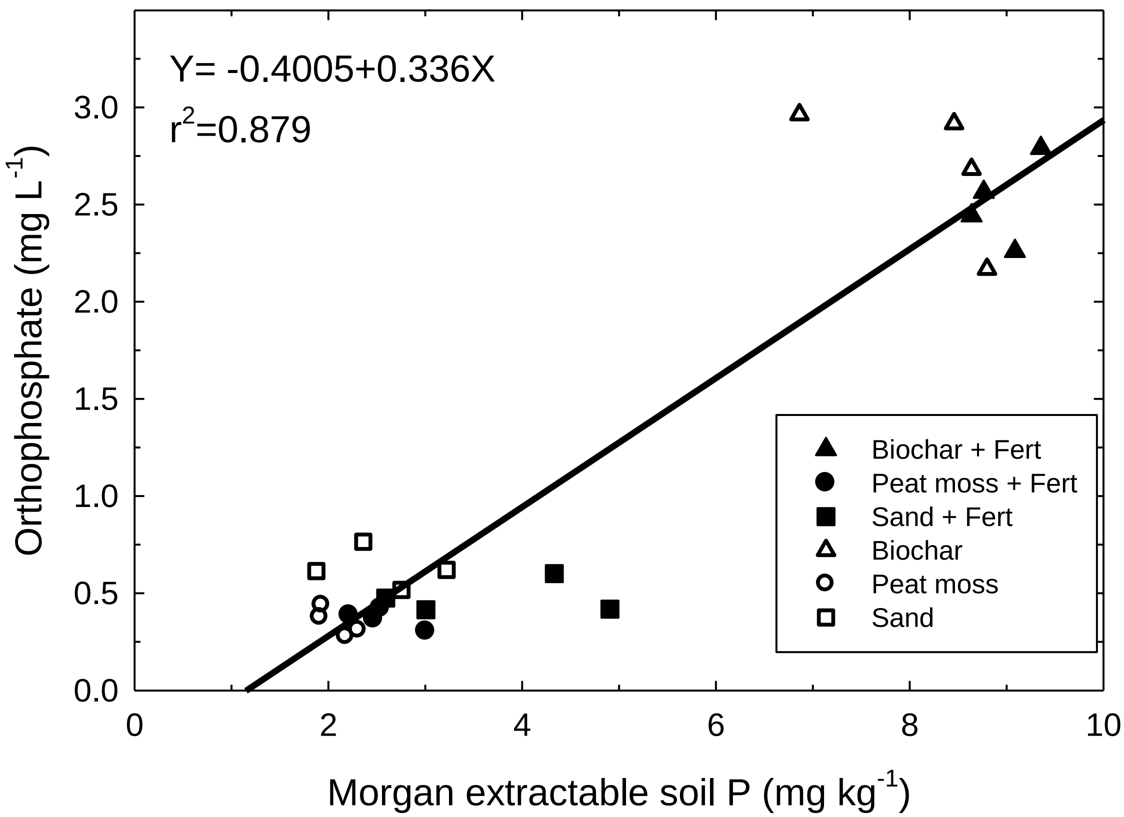

Figure 4-6. Relationship of orthophosphate P in leachate to Morgan extractable soil P from all treatment combinations. Leaching P concentrations are from the final leaching event in trial 2 (110 DAS). Soil cores were taken directly after the final leaching event to extract soil P levels.

There was a strong correlation among the amount of Morgan extractable phosphorus from the soil in the containers and the amount found in leachate. This provides an explanation for the higher amount of P in the leachate of the biochar containers being related to the available phosphorus pool in the soil. This may also explain the increased productivity or above ground density of the biochar amended pots. Phosphorus is a plant essential nutrient that is needed in larger quantities during establishment of turfgrass (Christians, 2004). Biochar provided more available P to the germinating turfgrass plants, allowing them to root and establish, while P was more limiting in the sand and peat moss amended containers. Under current nutrient guidelines provided through Cornell University for P fertilization, the Morgan P values found in the sand and peat moss containers would fall in the low range; while biochar amended container P values would be classified as adequate. It has also been hypothesized that the incorporation of biochar to soil or sand creates a colloidal fraction that is able to leach through the soil profile. These colloids may transport significant portions of bound phosphorus from the root zone compared to other organic sources.

Grass biochar appears to be a suitable organic amendment alternative that improves the quality and density of L-93 creeping bentgrass during establishment compared to plants grown in sand or peat moss amended sand root zones. The incorporation of biochar as a soil amendment may provide reduced fertilizer inputs by providing plant essential phosphorus and nitrogen, while improving nutrient availability to the establishing grasses. Further research should be conducted to develop relationships between nutrient additions attained by the incorporation of biochar, as well as the potential benefits of various parent materials for producing biochar for use in the turfgrass industry that incorporate a range of different biochar types with different nutrient loading potentials to optimize the nutrient benefits of biochar while reducing nutrient leaching potential.

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